FISHERY MANAGEMENT INVESTIGATIONS

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BONNER LAKE BURBOT STOCKING EVALUATION

ABSTRACT

The Kootenai Tribe of Idaho and the Idaho Department of Fish and Game developed a Burbot *Lota lota* hatchery supplementation program to increase abundance of Burbot in the Kootenai River system and restore angling opportunity. Excess hatchery production was available from 2013 through 2017, which allowed Burbot to be stocked in Bonner Lake. In 2018, we sampled Burbot in Bonner Lake to assess the effectiveness of the supplementation effort. We caught 2.7 (± 1.0; 80% C.I.) Burbot per net night in trammel nets. The majority (93%) of Burbot collected in our survey were assigned by parental based tagging to the 2015 year class. We found that Burbot length increased little since 2017. Our observations suggest that Burbot post-stocking survival was poor for most cohorts and growth was slow. We recommend continued annual monitoring to evaluate the influence of fish length and outplant timing on improving post-release survival of hatchery Burbot in Bonner Lake.

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INTRODUCTION

Burbot *Lota lota* are native to the Kootenai River drainage. Following construction of Libby Dam on the Kootenai River near Libby Montana, wild production of Burbot in the Idaho reach of the Kootenai River declined. In response, the Kootenai Tribe of Idaho and the Idaho Department of Fish and Game (IDFG) developed a hatchery supplementation program to increase abundance of Burbot in the system and restore angling opportunity.

Bonner Lake is located in Boundary County, Idaho, 14 km east of Bonners Ferry, Idaho. The 9.7-ha lake has a mean depth of 6.7 m and a maximum depth of 18 m. Bonner Lake is managed as a mixed species fishery. Rainbow Trout *Oncorhynchus mykiss* and kokanee *Oncorhynchus nerka* are stocked annually in the lake. A complement of warmwater fish species are also present including Largemouth Bass *Micropterus salmoides*, Yellow Perch *Perca flavescens*, and Pumpkinseed *Lepomis gibbosus*. Excess Burbot production from the Kootenai River hatchery program was available from 2013 through 2017. This allowed Burbot to be stocked in Bonner Lake from 2014 through 2017 in an effort to utilize surplus production and provide additional angling opportunity. Bonner Lake was selected as a stocking location primarily because of its location within the Kootenai drainage and potential to provide adequate over summer habitat for Burbot (i.e., 18 m max depth).

In 2018, we sampled Burbot in Bonner Lake to assess the effectiveness of the supplementation effort. Specifically, we looked to determine if Burbot stocked in Bonner Lake survived and grew at high enough rates to support a fishery.

METHODS

We sampled Burbot in Bonner Lake following ice-off on April 30, 2018. Burbot were sampled using sinking trammel nets set perpendicular to shore overnight at six randomly assigned locations (Table 1). Nets were configured with two outer panels of 25.4-cm multifilament mesh and a single 2.5-cm inner multifilament mesh panel. Trammel nets were 48.8 m long and 1.8 m tall. We measured relative abundance of Burbot in Bonner Lake as catch-per-net-night (CPUE).

All fish caught were measured to total length (mm). We described growth of release groups where possible by using the increase in mean annual total length-at-age relative to mean length of the cohort at stocking.

Burbot stocking events varied by time, age, and size at release (Table 2). Stocking success was evaluated by comparing the relative return of release groups. Parental-based tagging (PBT) or passive integrated transponder (PIT) tags assigned individual fish to brood year. The PBT evaluations were completed by removing a fin clip from each Burbot collected. Fin clips were stored on Whatman paper prior to analysis. Analysis was completed by the IDFG Eagle Fish Genetics Laboratory. Prior to stocking, half-duplex PIT tags were inserted in the abdominal cavity of some Burbot in in the 2013 and 2014 cohorts. All Burbot collected were scanned with a PIT tag reader upon collection. Detected PIT tags were referenced to a tagging database to assign individuals to brood year and stocking cohort.

RESULTS

We caught 16 Burbot among all nets (CPUE = 2.7 ± 1.0 fish/h; 80% CI). PBT analysis assigned 93% of Burbot caught to the 2015 year class (n = 15) and 7% to the 2014 year class (n = 1). Total length of Burbot caught varied from 278 to 411 mm. Mean total length of fish assigned to the 2015 year class was 312 mm. Length of the single fish representing the 2014 year class was 411 mm.

DISCUSSION

Our survey suggests that only hatchery-origin Burbot from the 2014 and 2015 cohorts survived in Bonner Lake. These results are consistent with a 2017 survey of Bonner Lake that only found Burbot representing the 2014 and 2015 year classes (Ryan et al. 2020c). Relative abundance of detected age classes declined from 2017, but the proportion of the population represented by these two age classes remained relatively constant. Ryan et al. (2020c) reported catching 9.3 Burbot per net in 2017. Burbot from the 2015 year class were also well represented in that survey, making up 98% of the catch. Although post-stocking survival has been low, it is apparent that Burbot can survive in Bonner Lake when stocked under suitable conditions. A number of factors may influence post-stocking survival, including size at release, timing of stocking, and water temperature at release. To date, opportunity to evaluate these type of variables has been limited due to inconsistency in the availability of Burbot. As such, we recommend continued monitoring of hatchery Burbot in Bonner Lake in an effort to better understand factors that influence survival. Conducting an annual survey requires minimal effort and additional data will better inform decisions about whether to continue these stocking efforts over the long-term.

Burbot growth in Bonner Lake appears to be slow, although our assessment should be interpreted with some caution because of low sample size. Mean length-at-age of Burbot from the 2015 year class that were caught in 2017 and 2018 surveys only increased by 34 mm between surveys. However, growth by stocking cohort was not clearly identifiable because three groups of Burbot from the 2015 year class were stocked in Bonner Lake. Stocking groups included two release years and two release seasons. Juvenile Burbot in the 2015 year class were not segregated by parent at the hatchery prior to release, prohibiting the identification of individuals within the 2015 year class to stocking group. In contrast, a single stocking group represented all Burbot of the 2014 year class. Total length of the single individual Burbot detected from the 2014 year class was equal to the estimated mean length of that year class in 2017 (i.e., 411 mm). Our observations of Burbot growth were not consistent with prior monitoring work conducted on Bonner Lake. Ryan et al. (2020c) suggested that Burbot grew rapidly initially after stocking. It is unclear why growth rate may have changed, but we speculate forage availability and or water temperature patterns may have played a role.

To date, the use of hatchery-origin Burbot in Bonner Lake has not produced a reliable fishery. Poor survival and growth of stocked Burbot appear to be limiting fishery potential. As such, we recommend discontinuing any emphasis placed on advertising this fishery as a new opportunity. Some stocked Burbot do survive, so continued attempts to build a fishery may be worthwhile if surplus Burbot are available and stocking costs are low. However, we recommend this be done opportunistically and not be viewed as a management priority.

RECOMMENDATIONS

1. Continue monitoring survival of hatchery Burbot in Bonner Lake.

2. Do not emphasize the use of hatchery Burbot in Bonner Lake as an angling opportunity unless survival improves.

3. Stock Burbot in Bonner Lake opportunistically (i.e., low management priority).

Table 1. Bonner Lake Burbot sampling locations from the 2018 survey.

Water	Date	Site	Latitude	Longitude
Bonner Lake	4/30/2018	1	48.727351	-116.110874
Bonner Lake	4/30/2018	2	48.726766	-116.109889
Bonner Lake	4/30/2018	3	48.726009	-116.110951
Bonner Lake	4/30/2018	4	48.724959	-116.109138
Bonner Lake	4/30/2018	5	48.724649	-116.106884
Bonner Lake	4/30/2018	6	48.723530	-116.106521

Table 2. Bonner Lake Burbot stocking history including year stocked (Year), year class (Class), date of release (Date), total number released (Released), and mean length of released individuals (TL).

Year	Class	Date	Released	TL (mm)
2014	2013	10/30/2014	18	224
2014	2014	10/30/2014	82	110
2015	2015	10/16/2015	276	90
2016	2015	9/8/2016	430	265
2016	2015	5/12/2016	1452	210
2016	2016	10/11/2016	1882	80
2017	2017	10/11/2017	1400	96
2017	2015	10/11/2017	200	386

KOOTENAI RIVER REDBAND TROUT INVENTORY

ABSTRACT

The distribution and abundance of interior Redband Trout *Oncorhynchus mykiss gairdneri* has been negatively impacted rangewide by a variety of factors. The Conservation Strategy for Interior Redband Trout in the States of California, Idaho, Montana, Nevada, Oregon, and Washington (Conservation Strategy) was completed as a collaborative multi-state effort to identify conservation priorities and guide conservation actions for the species rangewide. Multiple objectives were identified in the Conservation Strategy to improve and (or) update the understanding of Redband Trout occupancy and genetic status in the Kootenai River basin. In 2018, we completed an inventory of Redband Trout in the Idaho segment of the Kootenai River basin to address Conservation Strategy objectives by improving and (or) updating knowledge of Redband Trout distribution and genetic status. We widely sampled fish communities in Idaho tributaries to the Kootenai River to estimate species composition, density and collect tissue samples from *Oncorhynchus spp*. for the purpose of genetic evaluation. Genetic analysis identified species and species hybrids encountered, genetic diversity within populations, hybridization rates, intraspecific introgression of *O. mykiss* with coastal origin hatchery stocks, and interspecific introgression of *O. mykiss* and Westslope Cutthroat Trout *O. clarkii*. We found Redband Trout were distributed in open drainages where no known natural barrier to migration exists. In contrast, Westslope Cutthroat Trout were distributed in isolated stream reaches above natural barriers to migration. Core and conservation Redband Trout populations were identified in Boundary, Long Canyon, Ruby, Dodge, Trail, Twentymile, and Callahan creeks. Intra- and interspecific introgressive hybridization was evident in multiple tributaries, but hybrid zones were most common in the lowermost sampled reach of tributaries. Management opportunities to restore or improve native species distribution exist in isolated stream reaches where hybridization occurs and (or) where non-native species occur in isolation.

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INTRODUCTION

The distribution and abundance of interior Redband Trout *Oncorhynchus mykiss gairdneri* has been negatively impacted rangewide by a variety of factors (Muhlfeld et al. 2015). The Kootenai River basin (Kootenai Geographic Management Unit) of Idaho and Montana represents one portion of the native range of Redband Trout where distribution and abundance are thought to be reduced from historical levels (Muhlfeld et al. 2015). As a result, Kootenai Geographic Management Unit (GMU) Redband Trout were petitioned for listing under the Endangered Species Act in 1994. Listing was not granted due to insufficient availability of information to classify Redband Trout in the Kootenai GMU as a unique population segment.

The Conservation Strategy for Interior Redband Trout in the States of California, Idaho, Montana, Nevada, Oregon, and Washington (Conservation Strategy) was completed as a collaborative multi-state effort to identify conservation priorities and guide conservation actions for the species rangewide (IRCT 2016). The Conservation Strategy identifies a number of factors, both natural and anthropogenic, that likely influence the representation, resiliency, and redundancy of Redband Trout in the Kootenai GMU. For example, natural and unnatural barriers to migration may influence the distribution of Redband Trout among Kootenai River tributaries (Paragamian et al. 2008). In addition, interspecific and intraspecific (with coastal origin Rainbow Trout *O. m. irideus*) hybridization of Redband Trout has been described in the system and may reduce the viability of populations where it occurs (Knudsen et al. 2002, Paragamian et al. 2008, Williams and Jaworski 1995) .

Knowledge gaps in species distribution and genetic status were considered the primary limiting factor to implementation of actions aimed at improving representation, resiliency, and redundancy of Redband Trout in the Kootenai GMU (IRCT 2016). Species distribution and genetic status of native fishes in Kootenai GMU tributaries have been the focus of several investigations (Downs 1999, Fredericks and Hendricks 1997, Knudsen et al. 2002, Paragamian et al. 2008, Walters 2004, Walters et al. 2007, Williams and Jaworski 1995). However, uncertainty surrounding Redband Trout and Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* distributions in Idaho tributaries to the Kootenai River and their genetic status remain. As such, multiple objectives were identified in the Conservation Strategy to improve and or update the understanding of Redband Trout occupancy and genetic status in the basin.

In 2018, we completed an inventory of Redband Trout in the Idaho segment of the Kootenai GMU to expand our knowledge of Redband Trout distribution and genetic status. Specifically, this inventory described the genotypic distribution of *Oncorhynchus spp*. in tributaries of the Kootenai River in Idaho and estimated interspecific and intraspecific hybridization rates in Redband Trout and Westslope Cutthroat Trout populations. Information gained from this effort will be used to inform management actions aimed at conserving native fishes in the Kootenai GMU.

OBJECTIVES

1. Describe distribution and abundance of fish species occupying tributaries of the Kootenai River in Idaho.

2. Describe genotypic species distribution of Redband Trout, Rainbow Trout (i.e., coastal origin), and Westslope Cutthroat Trout.

3. Describe intraspecific and interspecific hybridization rates in Redband Trout and Westslope Cutthroat Trout.

4. Describe trends in hybridization rates in the basin where prior information exists.

5. Identify potential conservation actions that benefit native fishes in the Kootenai GMU.

METHODS

The 2018 Kootenai River GMU Redband Trout inventory was a collaborative effort among the Idaho Department of Fish and Game (IDFG), the Kootenai Tribe of Idaho (KTOI), and United States Forest Service (USFS). Project planning and implementation was led by IDFG with the assistance of other project partners. Field data collection, was implemented by representatives from IDFG and USFS from July through October 2018. Genetic analysis was implemented by the IDFG Eagle Fish Genetics Laboratory.

Fish Sampling

We widely sampled fish communities in Idaho tributaries to the Kootenai River. Selected streams included locations where fish distribution had not been described or where additional detail on fish distribution was beneficial. We included some additional locations where species distribution and hybridization rates have previously been described, but updated information provided a beneficial understanding of trends in distribution and hybridization rates.

Sample sites were predetermined using Arc GIS software (ESRI, Arc Map 10.6). We sampled one to three sample sites per stream (Table 3). Where multiple sites were sampled in a given tributary, sites were distributed uniformly throughout the stream in an effort to describe species distribution and variation in abundance. Access to the upstream extent of most tributaries was difficult. Although sample sites were distributed throughout some tributaries, the uppermost sample sites in most cases did not represent the full extent of fish habitat.

Fish were sampled with backpack electrofishing units. Settings varied among sites due to differences in water conductivity, but generally included 60 Hz and 700 to 800 volts. Most sites were sampled by two people, one person operating the electrofisher and one netting fish. All fish collected were identified, measured (total length, mm) and released downstream of the sample transect. We closed sample sections using block nets at the downstream end of a survey section at most sites to prevent escapement during downstream electrofishing passes. Block nets were not used at several sites due to high flows or the creek being too wide for the net. At sites where no block net was used, we sampled in an upstream direction and used a natural stream feature such as a cascade to limit upstream movement of fish during the sample. Sampling strategies included both multi-pass removal and single-pass samples to facilitate estimation of fish abundance and to collect genetic tissue samples. On multi-pass samples we completed sequential passes until captures of an individual pass were no more than 20% of the total capture by species for the sum of passes. Typically, two or three passes were completed. Sample transects were approximately 100 m in length. We measured each transect length and estimated average transect width to facilitate density estimation. Average width was estimated from a series of measured widths spaced at 10 to 20 m intervals throughout each transect.

Abundance of tributary fish populations was estimated using multi-pass removal estimates (Zippin 1958) in combination with single-pass samples. Single-pass sampling was used to increase the number of possible sample sites surveyed. We estimated abundances from single-pass samples by generating a multi-pass regression model of abundance based on first pass collections (Meyer and Schill 1999). A single model of abundance based on first-pass collections was developed and included sample data from all tributaries and all target species. Capture efficiencies were consistent among all tributaries and species providing support that model predictions were valid across these boundaries. Abundance estimates included fish ≥ 75 mm total length due to sampling efficiency considerations. We derived abundance estimates and associated 80% confidence intervals for two- and three-pass samples using calculations for removal estimates in closed populations (Hayes et. al 2007). We reported the total catch on the first pass as the population estimate when all the individuals of a particular species were captured on the first pass. In cases for which lower confidence bounds were less than the total number of fish captured, the total number of fish captured was reported as the lower bound. We reported density estimates as the number per 100 m^2. Mean density estimates by species and stream were estimated by averaging density estimates from all locations sampled in an individual drainage.

Genetic Analysis

Genetic analysis of *Oncorhynchus spp.* collected from Idaho tributaries to the Kootenai River was used to: 1) describe genotypic species distribution, 2) estimate interspecific and intraspecific hybridization rates of Redband Trout, coastal origin Rainbow Trout, and Westslope Cutthroat, and 3) describe trends in hybridization rates in the basin. Tissue samples were collected from the first 30 *Oncorhynchus spp.* at each sample site for genetic analysis, regardless of species or size. A small piece of fin tissue was removed from the caudal fin of each fish. Tissue samples were placed on white printer paper in paper coin envelopes for transport and storage prior to DNA extraction.

Hybridization was defined as the crossing of individuals from different taxa. Hybridization was described by individual sample sites in two ways: 1) as the percentage of identified hybrid fish within a sample of *Oncorhynchus* species (i.e., hybridization), and 2) as the proportion of Rainbow Trout and (or) Westslope Cutthroat Trout alleles within the total alleles identified (i.e., introgression). The direction of hybridization or introgression was based on the most prevalent taxa.

DNA were extracted from sampled fin tissue using nexttec™ Genomic DNA Isolation Kits from XpressBio (Thurmont, Maryland). All samples were genotyped at a panel of 379 single nucleotide polymorphism (SNP) loci using the Genotyping-in-Thousands Sequencing (GTseq) methodology developed by Campbell et al. (2015). This panel was developed by the Columbia River Inter-Tribal Fish Commission (CRITFC) and Idaho Department of Fish and Game (IDFG) to monitor *O. mykiss* populations throughout the Columbia River Basin (Steele et al. 2013, 2016; Hess et al. 2015). Within this panel, 16 loci exhibit diagnostic SNPs between Rainbow

Trout/Redband Trout and Westslope Cutthroat Trout (OMS00116_HYB, OMS00128_HYB, OMS00138_HYB, Ocl_gshpx357_HYB, Omy_104519624_HYB, Omy_10780634_HYB, Omy_130524160_HYB, Omy_GREB1_10_HYB, Omy_LDHB2_i6_HYB, Omy_RAD1454172_HYB, Omy_RAD4246532_HYB, Omy_RAD5540454_HYB, Omy_RAD7606020_HYB, Omy_g1282_HYB, Omy_metB138_HYB, Omy_nkef241_HYB). Genotypes from these 16 loci were input into the software program NewHybrids (Anderson and Thompson 2002) and used to estimate the probability that a sample belonged to one of six a posteriori categories: Rainbow Trout, Westslope Cutthroat Trout, F1 hybrids, Rainbow Trout backcrosses (i.e., hybrid spawning back to pure taxa) or Westslope Cutthroat Trout backcrosses. Individuals that did not assign to a single category at >90% were classified as "Other Hybrid".

Any samples assigned to a category other than RBT were removed from subsequent analyses. Redband Trout are native to the Kootenai River basin, but there is potential for hybridization/introgression with non-native coastal origin hatchery Rainbow Trout strains that have been stocked in the past to provide sport fishing opportunities. To evaluate intraspecific hybridization/introgression, we used reference sample collections of hatchery Rainbow Trout strains and the program STRUCTURE (v. 2.3.4 Pritchard et al. 2000) to estimate the inferred ancestry of each individual and sample collection admixture/introgression. We set a K = 2 populations where the two populations represent native Redband Trout versus non-native hatchery Rainbow Trout strains of "Coastal" origin. For our analyses, we included previously genotyped hatchery Rainbow Trout reference strain collections that had been screened with the same SNP marker panel: Fish Lake, Eagle Lake, Shasta, Arlee, Arlee x Erwin, and McConaughy. For each sample, STRUCTURE estimated the qi proportion of individual i's genome that originated from each of the two populations. Default parameters of admixture and correlated allele frequencies were used. The level of intraspecific introgression from hatchery Rainbow Trout was measured as the mean of hatchery coastal ancestry across individuals in each sample collection.

Expected heterozygosity (H_E) in each sample collected was estimated using the diveRsity package (Kennan et al. 2013) in R using the divBasic function (R Development Core Team, 2016). The diversity package was also used to calculate pairwise genetic differentiation among sample collections using the diffCalc function (F_{ST}; Weir and Cockerham 1984). To visualize genetic population structure and the influence of intraspecific hybridization/introgression from coastal hatchery rainbow trout we constructed a neighbor-joining (NJ) tree in the software program MEGA (Kumar et al 2018) from pairwise F_{ST} estimates.

Genotypes were stored on a Progeny (http://www.progenygenetics.com/) database server housed at Eagle Fish Genetics Laboratory. The final dataset used for this study may be made publically available at the standardized genetic repository www.fishgen.net.

In this investigation, our description of both phenotypic and genotypic species distribution were referenced to natural geologic features that are barriers to upstream fish movement where they occur. Our knowledge of barrier locations was obtained from a combination of IDFG ArcMap reference layers (personal communication, Evan Brown, Idaho Department of Fish and Game) and specific barrier surveys in the basin (Paragamian et al. 2008). Disagreement in barrier references existed. For example, agency reference layers indicated a natural barrier is present low in the Long Canyon Creek drainage. In contrast, Paragamian et al. (2008) specifically identified no barrier to fish migration was present in the lower segment of this drainage. Where reference information disagreed we used information obtained from formal barrier surveys (Paragamian et al. 2008).

We qualitatively compared our estimates of hybridization and introgression between *O. mykiss* and Westslope Cutthroat Trout to estimates from a 2006 genetic survey (Paragamian et al. 2008) to describe general trends in these rates. Sample sites in each survey were not replicates. In addition, strong variation in species presence as well as hybridization and introgression rates by location was apparent in our survey. As such, broad comparisons within or among tributaries were not deemed valid and precluded more rigorous quantitative comparisons. To facilitate comparisons, we identified sample sites in each survey that generally represented similar stream reaches and used these pairings to make comparisons by stream reach. Sample site pairings did not cross boundaries formed by migration barriers in individual streams as they represented strong differences in species composition in many locations in our survey.

RESULTS

Fish Sampling

We visited 77 sites among 36 tributary streams in the period from July 9 through October 4, 2018 (Table 3). We completed an electrofishing survey to estimate fish density at 67 of these sites encompassing 32 streams. Fish were detected at 62 of the 67 sites at which a survey was completed (Table 4). At 6 of the 77 sites visited, no estimate of fish density was made and only fish tissue samples for genetic analysis were collected. These sites were located near the confluence of their respective tributary and the Kootenai River. Four sites were visited, but either no water was present or stream size and water quantity prevented fish presence, so no survey was completed.

A single regression model was developed to estimate abundance for all single pass collections (Figure 1). Our capture efficiency in multi-pass samples was consistent (0.72 ± 0.19; mean ± SD) among tributaries and species, providing support that our model predictions were valid across these variables. Based on the developed linear model, our first-pass collections described approximately 80% of the variation in estimated abundance from multi-pass samples.

Phenotypic Rainbow Trout was found in 26 of the 32 tributaries surveyed, and was the most abundant fish species found in 17 of these streams (Table 4; Table 5; Figure 2). Densities varied from 0.1 to 36.0 fish/100 m^2. Mean total length of Rainbow Trout varied from 75 to 137 mm.

Westslope Cutthroat Trout were found in 11 tributaries with densities varying from 0.1 to 36.7 fish/100 m^2 (Table 4; Table 5; Figure 2). They were the dominant species in Ball, Burton, John Crown, Parker, Snow, and Trout creeks. Mean total length of sampled Westslope Cutthroat Trout varied from 75 to 182 mm.

Brook Trout *Salvelinus fontinalis* were detected in 15 tributaries, and their densities varied from 0.5 to 14.2 fish/100 m^2 (Table 4; Table 5; Figure 2). Brook Trout were the dominant species in Brush, Caribou, Cow, Fall, Myrtle, and Smith creekx. Mean total length of Brook Trout varied from 58 to 194 mm.

Several other species and species hybrids were detected in sampled tributaries, but were not widely distributed (Table 4; Table 5; Figure 2). We found Bull Trout *Salvelinus confluentus* in Fisher, Trout, and South Fork Callahan creeks. Bull Trout density in these streams varied from 0.1 to 1.1 fish/100 m^2. Length of Bull Trout caught in our surveys varied from 80 to 203 mm.

Mountain Whitefish *Prosopium williamsoni* were found in Ball, Boundary, and Fisher creeks at densities from 0.3 to 0.5 fish/100 m^2. Rainbow Trout x Westslope Cutthroat Trout hybrids were identified phenotypically in 10 tributaries, but were generally not abundant and varied in density from 0.2 to 4.3 fish/100 m^2. Longnose Dace *Rhinichthys cataractae*, Sculpin *Cottus spp*., Redside Shiner *Richardsonius balteatus*, Pumpkinseed *Lepomis gibbosus*, and Black Bullhead *Ameiurus melas* were also detected in some locations (Table 4).

Genetic Analysis

Oncorhynchus spp. were present and identified in genetic evaluations at 53 sample sites (Table 6; Figures 2-8). Genetic analysis indicated *O. mykiss* were dominant at 38 of those sites. Intraspecific introgression (i.e., coastal ancestry) in *O. mykiss* was low (i.e., ≤ 5.0%) in 24 of the 38 samples and represented eight tributaries, including Ball Creek, Boundary Creek, Dodge Creek - site 1, Fisher Creek, Long Canyon Creek, North Fork Callahan Creek, South Fork Callahan Creek, Trail Creek, and Twentymile Creek. A gradient of intraspecific introgression was evident in the remaining 14 samples where *O. mykiss* were detected and varied from 6% to 99%. Intraspecific introgression was not highly prevalent in some streams (e.g., Ruby Creek - site 1, Caribou Creek, Caboose Creek, lower Dodge Creek), but was at levels > 5%. In contrast, coastal influence was prevalent and samples resembled *O. m. irideus* in individuals from East Fork Boulder Creek, lower Boulder Creek - site 1, and Ruby Creek - site 2 at 96% to 99%. All sample sites where *O. mykiss* were dominant and intraspecific introgression was low were located in stream segments where no natural barrier to fish migration to the Kootenai River was known to exist. All sample sites where *O. mykiss* were dominant and intraspecific introgression was high (i.e., >90%) were isolated above known migration barriers. *O. mykiss* were present at lower sites on Parker, Myrtle, Burton, Fall, Brush, and Cabin creeks, but too few individuals were collected to evaluate intraspecific introgression.

Phylogenetic relationships generated using metrics of genetic differentiation confirmed patterns observed in Structure analysis (Figure 9). Specifically, reference collections (e.g., coastal rainbows) were highly differentiated, and populations that exhibited strong coastal influence clustered proximate to coastal hatchery trout reference samples. Samples from sites exhibiting strong coastal influence (i.e., Ruby Creek - site 2, East Fork Boulder Creek, and Boulder Creek - site 1) also clustered with reference hatchery trout samples. Other samples exhibiting moderate to high levels of intraspecific introgression were also closely clustered with reference hatchery trout samples. Geographically, most samples exhibiting strong coastal influence were located in tributaries upstream of Bonners Ferry. Ruby Creek – site 2 and Caribou Creek, which are both tributaries to Deep Creek, represented two deviations from this pattern of distribution. The influence of geographic proximity was also apparent in clustered samples from Boundary Creek and most tributaries of Deep Creek.

Expected heterozygosity varied from 0.13 to 0.29 across all *O. mykiss* dominant populations. Those populations exhibiting no or low (<5%) intraspecific introgression and where coastal ancestry was dominant (i.e., >90%) exhibited the lowest H_E (Figure 10).

Hybridization and introgression in samples where *O. mykiss* were dominant were generally low (Table 4; Figures 2-8). No hybridization with cutthroat trout was detected in 71% of those samples where intraspecific introgression of *O. mykiss* was ≤5%. Where hybridization was detected, 7% to 59% of our sample were hybrids. Interspecific introgression in these same samples was <5% in 20 of 24 samples. Where coastal ancestry was represented at levels >5%, only 3 of 14 samples exhibited hybridization levels >5%. Interspecific introgression within these same samples was not widespread at levels of ≤2% in 12 of the 14 samples. Across all *O. mykiss* dominant samples, interspecific introgression was generally most prevalent at the lowermost sample sites in a drainage. Specifically, interspecific introgression was estimated at 10% to 67% at lower drainage sites on Ball, Caboose, Caribou, and Fisher creeks.

Westslope Cutthroat Trout were most common in 12 of the 53 samples where *Oncorhynchus spp.* were detected (Table 6; Figures 2-8). Of these samples, only Blue Joe Creek (Boundary 8), Trout Creek (Trout – site 1 and Trout – site 3), and Boulder Creek (Boulder – site 3) exhibited hybridization with *O mykiss*. Where hybridization was detected, levels varied from 7% to 61%. Although hybridization of Westslope Cutthroat Trout dominated samples was not widely detected, Rainbow Trout alleles were detected in most samples. However, introgression levels were estimated at ≤5% in 10 of the 12 samples. All sample sites where Westslope Cutthroat Trout were most common were located above natural barriers that prevent upstream migration of fish from the Kootenai River. Most streams with dominant Westslope Cutthroat Trout populations were located downstream of Bonners Ferry. John Crown Creek is located upstream of Bonners Ferry and represented one exception to this pattern. Westslope Cutthroat Trout were abundant in John Crown Creek above a natural fish passage barrier.

No single species was common in samples from Burton Creek, Parker Creek – site 1, and Boulder Creek – site 2 (Table 6; Figures 2-8). All samples demonstrated high levels of hybridization between *O. mykiss* and Westslope Cutthroat Trout. Interspecific introgression at these locations varied from 33% to 43%.

Evidence of recent hybridization events was limited in most samples (Table 6). First-generation hybrids were detected at 10 of 53 locations. More distant hybridization events (e.g., F2, backcrosses) were more widely detected. Generally, hybridization events were most prevalent at locations representing the lowermost sample site in a tributary. For example, hybridization was 10% or greater at lower drainage sites in Ball, Caribou, Caboose, Fisher, Trout, and Parker creeks.

Sixteen sample sites from 11 tributaries represented comparable stream reaches between 2006 and 2018 genetic surveys for the purpose of evaluating trends in hybridization and introgression of *O. mykiss* and Westslope Cutthroat Trout (Table 7). Hybridization increased from 18.5% in 2006 to 36.7% in 2018 in lower Ball Creek. All other comparisons represented stable or decreasing rates between surveys. Interspecific introgression of *O. mykiss* increased only in lower Ball Creek from 4.1% to 66.6%. Introgression in the remaining stream reaches where *O. mykiss* was most common either decreased or was stable between surveys. Introgression of Westslope Cutthroat Trout was estimated to be greater in upper Ball, upper Boulder, upper Snow, and middle Trout creeks in 2018. However, differences in introgression rates between years were minimal (0.2% to 3.6%). Interspecific introgression in the remaining stream reaches where Westslope Cutthroat Trout were dominant either decreased or was stable between surveys.

DISCUSSION

Direct comparison of fish densities described in our survey to those from prior surveys was difficult due to variability in survey methods and abundance estimators through time. However, broad qualitative comparisons provide some perspective on population-level trends over time. Generally, we found density estimates were similar over time in tributaries where comparable estimates existed. For example, Fredericks et al. (1997) estimated *O. mykiss* densities (≥ age-1, ≥ 75 mm) varied from 2.0 to 15.5 fish/100 m^2 in select tributaries to Deep Creek, including Trail, Ruby, Dodge, Caribou, and Twentymile creeks. Our results were similar with *O. mykiss* densities varying from 2.9 to 22.5 fish/100 m^2 in these same tributaries. Although broad similarities existed, differences were also observed in other tributaries. We found few *O. mykiss* in lower Fall Creek (n = 2; no density estimated). In contrast, Fredericks et al. (1997) found *O. mykiss* were abundant in lower Fall Creek at 8.4 fish/100 m^2. Community-level species shifts were also detected in Fall Creek. We found only Brook Trout in sample sites above a known migration barrier. Walters et al. (2007) indicated upper Fall Creek was likely within the historic distribution of Westslope Cutthroat Trout and detected them in their survey. In Myrtle Creek, Paragamian et al. (2008) detected *O. mykiss* with strong coastal influences in mid-drainage genetic samples. In contrast, we found only Brook Trout in samples upstream of a known migration barrier low in the drainage.

Unique patterns of non-native species distribution and species presence that are not generally encountered in other portions of the Panhandle Region were represented in several Kootenai River tributaries. Specifically, Brook Trout were dominant and no other native species was detected in a majority of Fall and Myrtle creeks. Similarly, *O. mykiss* genetically resembling hatchery stocks were in East Fork Boulder and upper Ruby creeks with no other native species or sub-species detected. In addition, no fish were detected at sample sites above the Kootenai River floodplain in Katka and Caboose creeks. In all the listed locations, a known migration barrier exists below the stream reach where these conditions were observed. Brook Trout and Rainbow Trout have been widely stocked throughout the Panhandle Region. However, few examples in the Panhandle Region exist where native species (e.g., Westslope Cutthroat Trout) have been completely replaced or eliminated on a large stream reach scale (Ryan et al. 2020b; Ryan and Jakubowski 2012; Pend Oreille River Tributary Inventories, *see this report*). Ryan et al. (2014) found sink drainages flowing to the Rathdrum aquifer were occupied by only Brook Trout, but found no evidence that native fish previously occupied those streams or that any event occurred to remove native species. Similarly, we found no reference indicating the noted streams in our survey were previously occupied above migration barriers or that an event occurred that removed native fish assemblages. As such, we recommend cautious interpretation of references indicating these stream reaches represent portions of the native range of Redband Trout or Westslope Cutthroat Trout.

Our survey suggests Redband Trout and Westslope Cutthroat Trout were historically segregated throughout the Kootenai GMU in Idaho. We found *O. mykiss* were primarily distributed in tributaries and tributary reaches where no low elevation natural barriers to migration are known to occur. In contrast, Westslope Cutthroat Trout were primarily distributed in tributaries and tributary reaches above known natural barriers to fish migration. Our observations of distribution overlap with genotypic descriptions of core and conservation level Redband Trout and Westslope Cutthroat Trout populations in multiple tributaries, providing evidence these populations represent natural distributions rather than distribution resulting from management stocking efforts. Paragamian et al. (2008) described a similar species distribution in their investigation of species occurrence in the drainage. Other observations and hypotheses contribute to a similar understanding of species distribution in these drainages (Behnke 1992, Walters et al. 2007), cumulatively lending support to our interpretation. In contrast, Muhlfeld et al. (2015) and subsequently IRCT (2016) depicted a much broader historic distribution of Redband Trout that encompassed the majority of tributary habitat in the Idaho portion of the drainage. Their description of distribution was based in part on professional judgement rather than field-based observations. We recommend references to historical distribution be updated and incorporate field based investigations depicting a segregated distribution of Redband Trout and Westslope Cutthroat Trout based largely on presence of natural migration barriers in tributary streams.

Distribution of *O. mykiss* in lower Boulder, East Fork Boulder, and upper Ruby creeks did not follow the typical species distribution in the drainage. In these locations, *O. mykiss* were dominant in samples collected from sites above known migration barriers. *O. mykiss* in these samples exhibited strong coastal ancestry (i.e., ≥96%) suggesting populations were founded by hatchery stocks rather than native Redband Trout or mixed stocks. Given *O. mykiss* in these location so closely clustered with reference hatchery stocks, we speculate populations reflect historic introductions of hatchery fish into very low density native populations or fishless reaches of these respective tributaries. Stocking records corroborate *O. mykiss* were released in these streams, but little detail on timing or location of releases is available (unpublished data, Idaho Department of Fish and Game). We also found no record of large-scale events, such as piscicide treatment or stream dewatering that might have limited or removed existing native fish populations in these locations.

Genotypic descriptions of *O. mykiss* in our survey suggested Redband Trout are more widely distributed in lower Kootenai River tributaries than previously described and pure Redband Trout population segments are present. The Conservation Strategy identified three levels of genetic purity in its description of Redband Trout populations (IRCT 2016). These levels included core conservation populations (0% introgression), conservation populations (≤10% introgression), and sport fish populations (>10% introgression). Following that categorization, IRCT (2016) described three conservation Redband Trout populations in the Kootenai GMU located in Callahan Creek, Boundary Creek, and Twentymile Creek. Our results were consistent with their listing and described additional Redband Trout presence in drainages or segments of drainages, including Trail, Dodge, Ruby, and Long Canyon creeks. Low levels of intraspecific and interspecific introgression in Trail Creek Redband Trout suggest it is a conservation-level population. We found no evidence of introgression in Redband Trout sampled in upper Dodge Creek, suggesting a core–level population exists in the upper drainage. In contrast, intraspecific introgression low in the drainage has impacted genetic integrity and most likely represents a sport fish-level population in that portion of the drainage. A conservation population of Redband Trout was detected in lower Ruby Creek, but is likely impacted by an introduced coastal Rainbow Trout population in the upper drainage above a known migration barrier. In Long Canyon Creek, low levels of intraspecific introgression were detected in Redband Trout at our upper drainage sample site, suggesting a conservation population is present. Low in the Long Canyon Creek drainage, interspecific introgression of Redband Trout was moderate and hybridization was prevalent, suggesting this segment of the population may be more representative of a sport fish population. Similarly, we found segments of both the Boundary Creek and Callahan Creek drainages where no introgression was detected, suggesting core-conservation Redband Trout sources exist within these systems previously labeled as conservation populations.

Prior investigations that described intraspecific introgression of *O. mykiss* in Kootenai River tributaries are varied in their interpretation. For example, Williams and Jaworski (1995) described the presence of coastal alleles in Redband Trout in Long Canyon and Fisher creeks. In their work, they suggested the influence of introduced hatchery stocks was high and Redband Trout had largely been replaced in those drainages by coastal hatchery stocks. In contrast, Paragamian et al. (2008) found evidence of intraspecific introgression in these same streams, but suggested Redband Trout had not been completely replaced. Results from our investigation suggested intraspecific introgression in Long Canyon Creek is detectable, but levels were low (i.e., 2% to 5%) and *O. mykiss* in those streams represent Redband Trout. We did not evaluate the cause of discrepancies among surveys, but note several factors that may influence results in this type of survey, including evaluation method (i.e., gel electrophoresis vs. SNP panel), sample size and location, and (or) changes in the fish communities over time. In our investigation, we sampled multiple locations per stream that were distributed throughout a drainage when feasible.

We found evidence that species composition, hybridization, and introgression varied widely by sample location, both between and within tributaries. To address accuracy in describing species composition and genetic variation on a tributary scale, we recommend future surveys maximize the number of sample locations per tributary, especially when tributary features (e.g., habitat transitions) may influence fish distribution and movement.

Hybridization and interspecific introgression of *O. mykiss* and Westslope Cutthroat Trout were detected in our survey at moderate to high levels in some areas, but typically within predictable zones. These zones occurred low in most drainages at points of transition below migration barriers where dominant species differed above and below the barrier or in transitions of habitat where high gradient high quality habitats met low gradient disturbed habitats near the confluence of a tributary with the Kootenai River. We hypothesize our observations may be the result of downstream movement of fish within drainages in combination with exchange from fish utilizing the Kootenai River. Other abiotic influences may also impact zones of hybridization between *O. mykiss* and Westslope Cutthroat Trout. Young et al. (2016) found hybrid zones were significantly associated with warmer water temperatures, larger streams, proximity to warmer habitats and to recent sources of Rainbow Trout propagules. We observed an exception to this pattern in Trout Creek. Upper Trout Creek was dominated by Westslope Cutthroat Trout, but low level hybridization was detected. We hypothesize *O. mykiss* influence in this stream reach reflects a long history of stocking in Pyramid Lake, located at the headwaters of Trout Creek (unpublished data, Idaho Department of Fish and Game). Similarly, long and diverse stocking histories exist for both Boulder Creek and Cabin Creek, a tributary of Cow Creek (unpublished data, Idaho Department of Fish and Game). Brook Trout, *O. mykiss* and Westslope Cutthroat Trout were stocked in these drainages and likely are reflected in their current respective fish communities including a mix of *O. mykiss*, Westslope Cutthroat Trout, and their hybrids.

Results from Blue Joe Creek were unique relative to observations in the Boundary Creek drainage and elsewhere in the Kootenai GMU. Phenotypically, the dominant species at Blue Joe Creek – site 7 was *O. mykiss*. Westslope Cutthroat Trout were dominant upstream at Blue Joe Creek – site 8. Genetic evaluation generally supported field observations, but detected moderate hybridization and interspecific introgression at both sites. However, prior investigations utilizing environmental DNA (eDNA) suggested *O. mykiss* was dominant throughout the stream reach represented by these two sites (personal communication, Sean Stash, U.S. Forest Service). A *post hoc* evaluation indicated fish in this reach were primarily late-stage hybrids (e.g., > F1) with *O. mykiss* mitochondrial DNA and predominantly Westslope Cutthroat Trout nuclear DNA (personal communication, Matt Campbell, Idaho Department of Fish and Game). Thus, they appeared phenotypically as Cutthroat Trout, but did not amplify as Westslope Cutthroat Trout with an eDNA marker from a mitochondrial DNA gene. Upper Blue Joe Creek has a history of disturbance associated with mining activities in the upper drainage that have impacted the fish community in that reach (personal communication, Sean Stash, U.S. Forest Service). Other Boundary Creek samples identified Redband Trout as the dominant species present. No stocking record was found to indicate Westslope Cutthroat Trout were introduced to the upper drainage, but the strong presence of Redband Trout elsewhere in the drainage suggests stocking may have been the introductory mechanism.

Genetic diversity in *O. mykiss* populations described in our survey was low relative to other Redband populations in Idaho. We found H_E varied from 13 to 29%. In contrast, H_E from a wide array of Redband Trout populations in the upper Snake River basin of Idaho varied from 51% to 79% (Kozfkay et al. 2011). Redband Trout in our survey occurred in open segments of Kootenai River tributaries. In addition, evidence suggests migratory behaviors occur in Kootenai River Redband Trout (Fredericks et al. 1997, Downs 1999). Combined, these factors suggest within population isolation is not the likely cause of low diversity for Redband Trout. We speculate that observed genetic diversity is more likely an artifact of the geologic separation of the Kootenai River from the larger Upper Columbia River system. In contrast, *O. mykiss* representative of primarily coastal origins were found in isolated segments of investigated tributaries. We speculate these populations were founded by historic stocking events and do represent low genetic diversity as a result of population isolation. Alternatively, observed differences between our survey and that reported by Kozfkay et al. (2011), may be due to the method used to estimate H_E. They used microsatellites which can reach higher levels of H_E. In contrast, SNPs, which are commonly bi-allelic potentially limiting values of H_E, were used in our study.

Management Implications

Intra- and interspecific introgression of both Redband Trout and Westslope Cutthroat Trout in the Kootenai GMU is a concern relative to the conservation of both species (IRCT 2016, Paragamian et al. 2008). The native range of these species overlaps in the Kootenai GMU and hybridization was anticipated, as has been observed in other naturally sympatric populations (Kozfkay 2007). However, we found evidence that stocking, primarily of *O. mykiss irideus* (i.e., coastal origin), has influenced introgressive hybridization beyond that anticipated from natural events. As such, management of hybrid zones should be considered in an effort to reduce introgressive hybridization and conserve native stocks. Management opportunities will likely be limited where hybrid zones occur in open systems in close proximity to the Kootenai River as introgressive hybridization is high in these areas and they may represent naturally occurring hybrid zones. In contrast, where isolated zones of hybridization occur, management actions may be more feasible and should be prioritized (Appendix A). For example, intraspecific introgression may be reduced in Ruby Creek by removing existing *O. mykiss* with coastal origins that are isolated in the upper portion of that drainage. Other actions have already been taken to reduce the influence of non-native stocks. Specifically, stocking of *O. mykiss* in stream habitats within the Kootenai GMU no longer occurs and sterile Rainbow Trout are used in most mountain lake stocking efforts (unpublished data, Idaho Department of Fish and Game).

Redband Trout conservation strategies for Idaho tributaries of the Kootenai River should consider the influence of migratory life history within the system. We found Redband Trout occurred exclusively in open systems. Migratory behaviors have been documented in *O. mykiss* in these open systems (Fredericks et al. 1997, Downs 1999). For example, telemetry and tagging studies demonstrated large migratory *O. mykiss* captured in Deep Creek migrated to Kootenay Lake, British Columbia after spawning (Fredericks et al. 1997, Downs 1999). Preliminary investigations of *O. mykiss* origin and life history in the Kootenai River and tributaries using a microchemistry approach, identified both adfluvial and fluvial life history types (personal communication, T.J. Ross, Idaho Department of Fish and Game). The combination of these observations suggests migratory life histories do occur in Kootenai GMU Redband Trout. As such, populations may be influenced by habitat conditions and angler exploitation in spawning and rearing tributaries, the Kootenai River, and in Kootenay Lake. Although it is evident that migratory life histories do occur in Kootenai GMU Redband Trout, a broader understanding of migratory behaviors would be beneficial in forming management strategies for conservation and recreational fisheries.

Brook Trout expansion in Kootenai River tributaries is a concern relative to the conservation of native fish species. We found Brook Trout were widely distributed in the basin at varying densities. Brook Trout were the only species detected in isolated portions of Myrtle Creek and Fall Creek. Brook Trout in these drainages may have replaced other fish species, either native or non-native. For example, Rainbow Trout were previously described in overlapping portions of Myrtle Creek (Paragamian et al. 2008), but were not detected in middle and upper sample sites in our survey. Patterns of distribution described in this survey suggest these isolated locations most likely represent the historical distribution of Westslope Cutthroat Trout. These isolated stream segments where non-native fishes exist independently offer opportunities to restore or expand Westslope Cutthroat Trout distribution through removal and restocking efforts (Appendix A).

RECOMMENDATIONS

1. Cautiously interpret references to historic distribution of Redband Trout and (or) Westslope Cutthroat Trout in isolated segments of drainages in the Kootenai GMU where fish communities represent entirely non-native species or sub-species or where no fish are present.

2. Update references to historical distribution of Redband Trout and Westslope Cutthroat Trout in the Kootenai GMU to incorporate recent field-based investigations depicting a segregated distribution based largely on presence of natural migration barriers.

3. Use sampling strategies that maximize sampling effort throughout tributaries when describing species composition and genetic variation on a tributary scale.

4. Prioritize isolated zones of introgressive hybridization in targeted actions when seeking to improve genetic purity of Redband Trout and (or) Westslope Cutthroat Trout populations.

5. Continue use of sterile Rainbow Trout where stocking occurs as a tool for providing recreational fishing opportunities without risk of negative genetic outcomes to Redband Trout.

6. Continue investigations of migratory life histories to inform management strategies for the conservation of Redband Trout.

7. Consider opportunities to restore/introduce native trout in isolated stream reaches where non-native fishes currently exist (See Appendix A).

Table 3. Sample site locations and physical characteristics from electrofishing surveys conducted in 2018 on tributaries to the Kootenai River in Idaho.

Stream	Site ID	Date	Site Length (m)	Avg. Width (m)	Latitude	Longitude	Comment
Ball Creek	Ball 1	7/30/2018	100.0	7.4	48.794400	-116.420380	
	Ball 2	7/30/2018	97.4	8.3	48.796430	-116.499480	
	Ball 3	7/31/2018	104.9	4.6	48.772486	-116.597571	
Boulder Creek	Boulder 1	7/18/2018	100.0	8.6	48.598130	-116.092500	
	Boulder 2	8/2/2018	90.0	9.1	48.593790	-116.176930	
	Boulder 3	7/18/2018	100.0	7.0	48.555487	-116.229295	
	Genetics Only	10/4/2018	--	--	48.623456	-116.053970	No density estimate
Boundary Creek	Boundary 1	8/15/2018	94.3	12.9	48.995578	-116.571302	
	Boundary 2	8/22/2018	104.5	11.8	48.988420	-116.607800	
Shorty Creek	Boundary 3 - Shorty	8/9/2018	100.0	4.6	48.966406	-116.689653	
Grass Creek	Boundary 4 - Grass	9/11/2018	100.0	8.9	48.993240	-116.759560	
	Boundary 5 - Grass	9/11/2018	100.0	9.0	48.970640	-116.816810	
	Boundary 6 - Grass	9/13/2018	100.0	4.8	48.918570	-116.853761	
Blue Joe Creek	Boundary 7 - Blue Joe	9/12/2018	100.0	6.0	49.000140	-116.829310	
	Boundary 8 - Blue Joe	9/12/2018	100.0	7.0	48.957130	-116.876580	
Saddle Creek	Boundary 9 - Saddle	8/10/2018	100.0	4.7	48.969440	-116.719650	
Brown Creek	Brown 1	--	--	--	48.613523	-116.390153	No water
	Brown 2	--	--	--	48.601688	-116.375143	No water
	Brown 3	--	--	--	48.608771	-116.310169	No water
Brush Creek	Brush 1	7/9/2018	100.0	1.6	48.662011	-116.226008	
Burton Creek	Burton 1	8/9/2018	45.7	2.2	48.779090	-116.416740	
Cabin Creek	Cabin Creek 1	8/15/2018	104.0	1.9	48.660040	-116.242260	
Caboose Creek	Caboose 1	7/23/2018	89.9	2.0	48.647922	-116.098262	No fish
	Caboose 1A	8/1/2018	102.5	2.4	48.657930	-116.086710	No fish
	Genetics Only	10/4/2018	--	--	48.658180	-116.087000	No density estimate
Caribou Creek	Caribou 1	7/18/2018	105.0	6.0	48.658960	-116.404490	
	Caribou 2	7/18/2018	100.0	4.3	48.650380	-116.556900	

Table 3 (continued)

Stream	Site ID	Date	Site Length (m)	Avg. Width (m)	Latitude	Longitude	Comment
Cone Creek	Cone Creek	8/14/2018	101.5	2.8	48.539940	-116.314830	
Cow Creek	Cow 1	7/19/2018	100.0	2.7	48.693730	-116.255330	
Curley Creek	Curley	10/4/2018	--	--	48.648130	-116.069820	No density estimate
Dodge Creek	Dodge 1	7/17/2018	106.3	3.3	48.539399	-116.479817	
	Dodge Lower	7/19/2018	103.0	2.8	48.526020	-116.463680	
East Fork Boulder Creek	EF Boulder 1	7/31/2018	105.0	5.1	48.584790	-116.117390	
	EF Boulder 2	7/31/2018	100.0	6.0	48.562500	-116.124880	
	EF Boulder 3	7/31/2018	100.0	4.3	48.541780	-116.137430	
Fall Creek	Fall 1	7/17/2018	--	--	48.583090	-116.417070	No density estimate
	Fall 2	7/17/2018	97.5	6.7	48.577946	-116.503554	
	Fall 3	7/17/2018	99.0	1.6	48.604366	-116.537448	
Fisher Creek	Fisher 1	8/22/2018	97.4	2.9	48.882523	-116.440511	
John Crown/Debt Creek	John Crown 1	8/14/2018	109.3	1.5	48.663370	-116.112440	
	Genetics Only	10/4/2018	--	--	48.675420	-116.103070	No density estimate
Katka Creek	Katka 1	7/23/2018	100.0	2.4	48.684687	-116.150115	No Fish
	Katka 2	7/23/2018	64.0	1.5	48.660273	-116.176656	No Fish
	Genetics Only	10/4/2018	--	--	48.688660	-116.135060	No density estimate
Long Canyon Creek	Long Canyon 1	7/23/2018	100.0	11.0	48.950130	-116.536200	
	Long Canyon 2	8/8/2018	89.4	8.3	48.925230	-116.562650	
Lost Creek	Lost 1	8/22/2018	94.5	2.0	48.748229	-116.420746	
Marsh Creek	Marsh Creek	9/20/2018	100.0	1.9	48.951170	-116.792550	
Moyie River	Genetics Only	10/4/2018	--	--	48.719310	-116.187590	No density estimate
Myrtle Creek	Myrtle 1	7/26/2018	98.0	8.6	48.707560	-116.416530	
	Myrtle 2	7/26/2018	101.8	9.6	48.723254	-116.534549	
	Myrtle 3	7/24/2018	94.3	6.0	48.723114	-116.617215	
North Fork Callahan Creek	NF Callahan 1	7/24/2018	97.2	6.6	48.453930	-116.094130	
	NF Callahan 2	7/24/2018	94.5	9.1	48.464327	-116.124118	
Parker Creek	Parker 1	7/26/2018	99.0	5.2	48.918960	-116.492370	
	Parker 2	7/30/2018	88.3	5.9	48.906400	-116.506230	

Table 3 (continued)

Stream	Site ID	Date	Site Length (m)	Avg. Width (m)	Latitude	Longitude	Comment
Ruby Creek	Ruby 1	7/18/2018	103.0	4.2	48.619540	-116.401490	
	Ruby 2	7/19/2018	101.0	5.6	48.631710	-116.467850	
	Ruby 3	---	--	--	48.633555	-116.534657	No fish/No survey - too small
South Fork Callahan Creek	SF Callahan 1	7/24/2018	97.5	8.9	48.420250	-116.031050	
	SF Callahan 2	7/24/2018	101.0	6.8	48.399240	-116.080670	
Smith Creek	Smith 1	7/23/2018	110.0	11.8	48.962084	-116.554092	
	Smith 2	7/23/2018	100.0	22.7	48.934730	-116.648460	
	Smith 3	7/25/2018	102.6	7.1	48.919190	-116.705540	
Snow Creek	Snow 1	7/18/2018	100.0	7.8	48.667340	-116.425860	
	Snow 2	7/24/2018	100.0	7.3	48.685600	-116.470790	
	Snow 3	7/18/2018	110.2	4.7	48.687916	-116.542017	
Trail Creek	Trail 1	7/16/2018	100.0	3.8	48.568930	-116.387780	
	Trail 2	7/19/2018	45.7	3.5	48.553130	-116.360770	
	Trail 3	7/25/2018	60.0	1.4	48.532078	-116.342987	No Fish
Trout Creek	Trout 1	7/25/2018	104.0	8.1	48.832690	-116.432210	
	Trout 2	7/25/2018	100.0	7.9	48.823280	-116.525540	
	Trout 3	7/25/2018	100.0	4.6	48.803120	-116.600480	
Twenty Mile Creek	Twentymile 1	7/16/2018	100.0	2.6	48.586970	-116.381920	
	Twentymile 2	7/16/2018	96.5	3.4	48.586320	-116.332530	
	Twentymile 3	7/18/2018	95.0	3.9	48.575750	-116.289080	

Table 4. Survey results by stream, sampled site, and species for Kootenai River tributaries sampled in 2018. Catch (n) and length distributions (mean; minimum and maximum total length – TL) include fish of all lengths (mm), while only fish ≥ 75 mm were included in abundance estimates (Est. N).

Stream	Site	Species	n	Mean TL	Min-Max TL	Est. N	80% CI-	80% CI +	Fish/100 m^2
Ball Creek	1	BKT	1	69	69-69	--	--	--	--
Ball Creek	1	LND	20	95	71-113	21	20.0	23.5	2.9
Ball Creek	1	MWF	3	85	74-92	3	--	--	0.4
Ball Creek	1	RBT	26	97	71-165	27	26.0	28.8	3.6
Ball Creek	1	RXY	6	93	80-104	6	--	--	0.8
Ball Creek	1	SCL	2	84	38-91	2	--	--	0.3
Ball Creek	1	WCT	6	120	83-189	8	6.0	12.1	1.0
Ball Creek	2	RBT	1	61	61-61	--	--	--	--
Ball Creek	2	WCT	38	129	66-176	58	48.6	66.9	7.2
Ball Creek	3	WCT	56	139	54-202	85	75.7	94.6	17.8
Boulder Creek	1	RBT	13	129	71-211	20	13.0	28.8	2.3
Boulder Creek	1	RXY	1	94	94-94	2	1.0	10.6	0.2
Boulder Creek	2	BKT	9	--	--	10	9.0	11.0	1.2
Boulder Creek	2	RBT	47	134	92-211	48	47.0	48.7	5.8
Boulder Creek	2	RXY	24	216	203-228	24	24.0	25.0	3.0
Boulder Creek	2	WCT	28	136	96-180	33	28.0	39.6	4.0
Boulder Creek	3	BKT	7	--	--	11	7.0	19.7	1.5
Boulder Creek	3	RXY	2	120	83-156	3	2.0	12.1	0.4
Boulder Creek	3	WCT	14	137	69-206	21	14.0	30.3	3.0
Boulder Creek	Genetics	--	--	--	--	--	--	--	--
Boundary Creek	1	LND	11	93	46-131	17	11.0	25.8	1.4
Boundary Creek	1	MWF	4	86	71-100	6	4.0	15.2	0.5
Boundary Creek	1	PMK	3	--	61-65	--	--	--	--
Boundary Creek	1	RBT	31	105	71-159	47	38.1	56.2	3.9
Boundary Creek	2	LND	13	127	103-160	20	13.0	28.8	1.6
Boundary Creek	2	MWF	1	187	187-187	2	1.0	10.6	0.1
Boundary Creek	2	RBT	50	115	65-180	76	66.7	85.3	6.2
Boundary Creek	3 - Shorty	RBT	8	142	85-218	9	8.0	11.7	2.0
Boundary Creek	4 - Grass	BKT	1	--	--	2	1.0	10.6	0.2
Boundary Creek	4 - Grass	RBT	50	109	82-168	76	66.7	85.3	8.5
Boundary Creek	4 - Grass	RXY	3	85	81-88	5	3.0	13.6	0.5
Boundary Creek	5 - Grass	BKT	3	--	58-66	5	3.0	13.6	0.5
Boundary Creek	5 - Grass	RBT	31	110	46-158	47	38.1	56.2	5.2
Boundary Creek	6- Grass	BKT	4	--	--	6	4.0	15.2	1.3

Table 4 (continued)

Stream	Site	Species	n	Mean TL	Min-Max TL	Est. N	80% CI-	80% CI +	Fish/100 m^2
Boundary Creek	6 - Grass	RBT	11	119	74-160	17	11.0	25.8	3.5
Boundary Creek	7 - Blue Joe	BKT	2	--	--	3	2.0	12.1	0.5
Boundary Creek	7 - Blue Joe	RBT	35	157	96-222	53	44.1	62.3	8.8
Boundary Creek	7 - Blue Joe	RXY	1	240	240-240	2	1.0	10.6	0.3
Boundary Creek	8 - Blue Joe	WCT	10	158	118-212	15	10.0	24.2	2.2
Boundary Creek	9 - Saddle	RBT	52	148	87-217	79	69.7	88.4	16.7
Boundary Creek	10 - Marsh Creek	RBT	21	123	60-209	32	22.9	40.9	16.5
Brown Creek	1	No Survey	0	--	--	--	--	--	--
Brown Creek	2	No Survey	0	--	--	--	--	--	--
Brown Creek	3	No Survey	0	--	--	--	--	--	--
Brush Creek	1	BKT	5	194	155-220	5	5.0	6.6	3.3
Brush Creek	1	RBT	1	123	123-123	1	1.0	1.0	0.6
Burton Creek	1	BKT	1	110	110-110	2	1.0	10.6	1.5
Burton Creek	1	RBT	1	90	90-90	2	1.0	10.6	1.5
Burton Creek	1	SCL	1	77	69-77	2	1.0	10.6	1.5
Burton Creek	1	WCT	2	97	95-98	3	2.0	12.1	3.0
Cabin Creek	1	BKT	2	124	64-136	3	2.0	12.1	1.5
Cabin Creek	1	RBT	1	132	132-132	2	1.0	10.6	0.8
Cabin Creek	1	RXY	2	137	137-137	3	2.0	12.1	1.5
Caboose Creek	1	No Fish	0	--	--	--	--	--	--
Caboose Creek	1a	No Fish	0	--	--	--	--	--	--
Caboose Creek	Genetics	--	--	--	--	--	--	--	--
Caribou Creek	1	BKT		64	58-68	--	--	--	--
Caribou Creek	1	LND	17	84	53-104	26	17.0	34.9	4.1
Caribou Creek	1	MWF	3	67	61-70	--	--	--	--
Caribou Creek	1	RBT	12	98	73-130	18	12.0	27.3	2.9
Caribou Creek	1	RXY	1	108	108-108	2	1.0	10.6	0.2
Caribou Creek	1	SCL	1	75	68-75	2	1.0	10.6	0.2
Caribou Creek	2	BKT	28	120	58-161	43	33.5	51.6	9.9
Cow Creek	1	BKT	7	154	55-248	8	7.0	10.7	2.9
Curly Creek	Genetics	--	--	--	--	--	--	--	--
Dodge Creek	1	RBT	41	125	55-156	42	41.0	43.0	11.7
Dodge Creek	Lower	BKT	13	120	30-166	20	13.0	28.8	6.9
Dodge Creek	Lower	RBT	19	94	28-131	29	19.9	37.9	10.1
Dodge Creek	Lower	SCL	1	92	92-92	2	1.0	10.6	0.5
EF Boulder Creek	1	BKT	4	--	--	6	4.0	15.2	1.1
EF Boulder Creek	1	RBT	50	119	78-188	76	66.7	85.3	14.1
EF Boulder Creek	1	RXY	2	162	162-162	3	2.0	12.1	0.6

Table 4 (continued)

Stream	Site	Species	n	Mean TL	Min-Max TL	Est. N	80% CI-	80% CI +	Fish/100 m^2
EF Boulder Creek	2	BKT	3	--	--	5	3.0	13.6	0.8
EF Boulder Creek	2	RXY	1	118	118-118	2	1.0	10.6	0.3
EF Boulder Creek	3	BKT	8	--	--	8	8.0	8.9	1.9
EF Boulder Creek	3	RBT	41	100	53-225	45	41.0	49.0	10.4
Fall Creek	1	BKT	3	106	32-125	5	3.0	13.6	--
Fall Creek	1	LND	11	89	40-104	--	--	--	--
Fall Creek	1	MWF	1	129	73-73	--	--	--	--
Fall Creek	1	PMK	1	--	60-129	--	--	--	--
Fall Creek	1	RBT	2	111	111-111	3	2.0	12.1	--
Fall Creek	1	RSS	3	82	79-84	--	--	--	--
Fall Creek	1	SCL	3	83	55-90	--	--	--	--
Fall Creek	2	BKT	26	143	84-237	40	30.5	48.6	6.1
Fall Creek	2	RBT	1	37	37-37	--	--	--	--
Fall Creek	3	BKT	34	117	28-183	35	34.0	36.9	22.4
Fisher Creek	1	BLT	1	80	80-80	2	1.0	10.6	0.5
Fisher Creek	1	MWF	1	89	89-89	2	1.0	10.6	0.5
Fisher Creek	1	RBT	15	124	74-167	23	15.0	31.8	8.0
Fisher Creek	1	WCT	4	113	85-128	--	--	--	--
John Crown Creek	Genetics	--	--	--	--	--	--	--	--
John Crown Creek	1	WCT	16	107	31-214	24	16.0	33.3	14.6
Katka Creek	1	No Fish	0	--	--	--	--	--	--
Katka Creek	2	No Fish	0	--	--	--	--	--	--
Katka Creek	Genetics	--	--	--	--	--	--	--	--
Long Canyon Creek	1	BKT	4	152	102-180	4	4.0	5.7	0.4
Long Canyon Creek	1	LND	19	102	71-155	--	--	--	--
Long Canyon Creek	1	RBT	31	96	57-142	33	31.0	36.9	3.0
Long Canyon Creek	1	WCT	1	182	182-182	1	--	--	0.1
Long Canyon Creek	2	BKT	8	155	142-171	12	8.0	21.2	1.6
Long Canyon Creek	2	RBT	27	143	71-192	41	32.0	50.1	5.5
Lost Creek	1	RBT	1	120	120-120	2	1.0	10.6	0.8
Moyie River	Genetics	--	--	--	--	--	--	--	--
Myrtle Creek	1	BKT	1	45	45-45	--	--	--	--
Myrtle Creek	1	LND	28	94	47-124	--	--	--	--
Myrtle Creek	1	MWF	3	58	52-62	--	--	--	--
Myrtle Creek	1	RBT	3	114	104-121	5	3.0	13.6	0.5
Myrtle Creek	1	SCL	8	84	79-91	--	--	--	--
Myrtle Creek	1	WCT	1	75	75-75	2	1.0	10.6	0.2
Myrtle Creek	2	BKT	39	148	85-212	39	39.0	39.6	4.0

Table 4 (continued)

Stream	Site	Species	n	Mean TL	Min-Max TL	Est. N	80% CI-	80% CI +	Fish/100 m^2
Myrtle Creek	3	BKT	38	128	75-192	58	48.6	66.9	10.3
NF Callahan	1	RBT	88	123	68-192	116	94.8	136.4	18.1
NF Callahan	2	RBT	129	125	60-186	196	183.9	208.4	22.9
Parker Creek	1	RBT	3	137	128-154	5	3.0	13.6	0.9
Parker Creek	1	WCT	16	136	93-189	24	16.0	33.3	4.7
Parker Creek	2	WCT	72	135	65-228	76	72.0	79.8	14.5
Ruby Creek	1	LND	18	90	43-121	--	--	--	--
Ruby Creek	1	MWF	3	64	56-72	--	--	--	--
Ruby Creek	1	RBT	22	110	38-193	33	24.4	42.5	7.8
Ruby Creek	2	RBT	12	135	80-172	12	12.0	12.6	2.2
Ruby Creek	3	No Survey	0	--	--	--	--	--	--
SF Callahan Creek	1	BLT	11	109	90-202	17	11.0	25.8	1.9
SF Callahan Creek	1	RBT	60	125	57-226	91	81.7	100.8	10.5
SF Callahan Creek	2	BLT	1	203	203-203	2	1.0	10.6	0.2
SF Callahan Creek	2	RBT	63	123	50-186	96	86.2	105.4	13.9
Smith Creek	1	BBH	1	102	102-102	--	--	--	--
Smith Creek	1	BKT	1	59	59-59	--	--	--	--
Smith Creek	1	LND	11	95	53-126	--	--	--	--
Smith Creek	1	SCL	8	82	55-94	--	--	--	--
Smith Creek	2	BKT	3	150	112-179	5	3.0	13.6	3.6
Smith Creek	3	BKT	17	137	45-180	26	17.0	34.9	0.8
Smith Creek	3	WCT	4	163	126-206	6	4.0	15.2	0.4
Snow Creek	1	BKT	2	118	104-131	3	2.0	12.1	3.9
Snow Creek	1	WCT	20	144	48-243	30	21.4	39.4	1.3
Snow Creek	2	BKT	9	134	36-181	9	9.0	9.8	0.6
Snow Creek	2	RXY	4	111	79-148	4	4.0	4.0	4.3
Snow Creek	2	WCT	30	128	72-186	31	30.0	33.4	3.2
Snow Creek	3	BKT	11	138	83-173	17	11.0	25.8	3.8
Snow Creek	3	WCT	13	137	71-185	20	13.0	28.8	8.0
Trail Creek	1	LND	10	93	72-112	--	--	--	--
Trail Creek	1	RBT	44	102	32-160	67	57.7	76.1	17.8
Trail Creek	2	RBT	30	109	26-181	46	36.5	54.7	28.4
Trail Creek	3	No Fish	0	--	--	--	--	--	--
Trail Creek	Cone Creek	BKT	12	58	49-66	--	--	--	--
Trail Creek	Cone Creek	RBT	15	122	39-187	23	15.0	31.8	3.2
Trail Creek	Cone Creek	RXY	6	102	51-123	9	6.0	18.2	4.0
Trout Creek	1	BLT	1	134	134-134	1	--	--	0.1

Table 4 (continued)

Stream	Site	Species	n	Mean TL	Min-Max TL	Est. N	80% CI-	80% CI +	Fish/100 m^2
Trout Creek	1	RBT	2	118	57-118	1	1.0	1.0	0.1
Trout Creek	1	RXY	8	132	81-157	8	8.0	8.9	1.0
Trout Creek	1	WCT	24	128	66-204	25	24.0	25.7	2.9
Trout Creek	2	WCT	83	122	82-177	126	115.9	136.4	15.9
Trout Creek	3	WCT	110	127	67-196	167	155.9	178.6	36.7
Twentymile Creek	1	RBT	62	106	28-186	94	84.7	103.9	36.0
Twentymile Creek	2	RBT	49	112	48-188	50	49.0	51.7	15.4
Twentymile Creek	3	RBT	39	115	47-163	59	50.1	68.4	16.1

BBH = Black Bullhead
BKT = Brook Trout
BLT = Bull Trout
LND = Longnose Dace
MWF = Mountain Whitefish
PMK = Pumpkinseed
RBT = Rainbow Trout (O. mykiss)
RSS = Redside Shiner
RXY = Rainbow Trout x Westslope Cutthroat Trout Hybrid
SCL = Sculpin spp.
WCT = Westslope Cutthroat Trout

Table 5. Mean density (fish/100 m^2) of sampled salmonids from tributaries to the Kootenai River, Idaho in 2018. Density estimates represent only fish ≥ 75 mm. Mean density values were calculated by species for all surveyed sections within a stream drainage.

	Average of Fish/100 m^2					
Stream	BKT	BLT	MWF	RBT	RXY	WCT
Ball Creek	0.0	0.0	0.4	3.6	0.8	8.7
Boulder Creek	1.3	0.0	0.0	4.0	1.2	3.5
Boundary Creek	0.6	0.0	0.3	7.9	0.4	2.2
Brush Creek	3.3	0.0	0.0	0.6	0.0	0.0
Burton Creek	1.5	0.0	0.0	1.5	0.0	3.0
Cabin Creek	1.5	0.0	0.0	0.8	1.5	0.0
Caboose Creek	0.0	0.0	0.0	0.0	0.0	0.0
Caribou Creek	9.9	0.0	0.0	2.9	0.2	0.0
Cow Creek	2.9	0.0	0.0	0.0	0.0	0.0
Dodge Creek	6.9	0.0	0.0	10.9	0.0	0.0
EF Boulder Creek	1.3	0.0	0.0	11.4	0.4	0.0
Fall Creek	14.2	0.0	0.0	0.0	0.0	0.0
Fisher Creek	0.0	0.5	0.5	8.0	0.0	0.0
John Crown Creek	0.0	0.0	0.0	0.0	0.0	14.6
Katka Creek	0.0	0.0	0.0	0.0	0.0	0.0
Long Canyon Creek	1.0	0.0	0.0	4.3	0.0	0.1
Lost Creek	0.0	0.0	0.0	0.8	0.0	0.0
Myrtle Creek	7.1	0.0	0.0	0.5	0.0	0.2
NF Callahan Creek	0.0	0.0	0.0	20.5	0.0	0.0
Parker Creek	0.0	0.0	0.0	0.9	0.0	9.6
Ruby Creek	0.0	0.0	0.0	5.0	0.0	0.0
SF Callahan Creek	0.0	1.1	0.0	12.2	0.0	0.0
Smith Creek	1.9	0.0	0.0	0.0	0.0	0.8
Snow Creek	1.6	0.0	0.0	0.0	0.6	4.0
Trail Creek	0.0	0.0	0.0	18.1	3.2	0.0
Trout Creek	0.0	0.1	0.0	0.1	1.0	18.5
Twentymile Creek	0.0	0.0	0.0	22.5	0.0	0.0

BKT = Brook Trout
BLT = Bull Trout
MWF = Mountain Whitefish
RBT = Rainbow Trout
WCT = Westslope Cutthroat Trout
RXY = Rainbow x Cutthroat Trout Hybrid

Table 6. Results of genetic analysis from samples (N) of *Oncorhynchus spp.* collected from Idaho tributaries to the Kootenai River in 2018. Results identify species, expected heterozygosity (H_E), hybridization rate, intraspecific hybridization rate (of O. mykiss), and interspecific hybridization rate.

Stream	Site	N	O. Mykiss	O. Clarkii	F 1	F 2	Rainbow _BC	Cutthroat _BC	Other _HYB	HE	Hybridization	Intraspecific Introgression	Interspecific Introgression	Total alleles
Ball Creek	1	30	14	5	7	1	2	0	1	0.22	0.37	0.03	0.67	940
Ball Creek	2	29	0	29	0	0	0	0	0	N/A	0.00	N/A	0.00	926
Ball Creek	3	31	0	31	0	0	0	0	0	N/A	0.00	N/A	0.01	990
Boulder Creek	1	29	27	0	0	0	2	0	0	0.23	0.07	0.99	0.02	954
Boulder Creek	2	30	0	27	1	0	0	2	0	N/A	0.10	N/A	0.05	954
Boulder Creek	3	30	3	4	2	5	8	2	6	N/A	0.77	N/A	0.43	944
Boulder Creek	Genetics	30	30	0	0	0	0	0	0	0.26	0.00	0.12	0.00	960
Boundary Creek	1	30	28	0	1	0	1	0	0	0.23	0.07	0.04	0.00	954
Boundary Creek	2	30	30	0	0	0	0	0	0	0.24	0.00	0.05	0.00	958
Boundary Creek	3 - Shorty	27	27	0	0	0	0	0	0	0.08	0.00	0.00	0.00	864
Boundary Creek	4 - Grass	30	30	0	0	0	0	0	0	0.21	0.00	0.02	0.00	960
Boundary Creek	5 - Grass	30	30	0	0	0	0	0	0	0.21	0.00	0.02	0.00	960
Boundary Creek	6 - Grass	29	29	0	0	0	0	0	0	0.20	0.00	0.01	0.00	922
Boundary Creek	7 - Blue Joe	30	26	0	2	0	1	0	1	0.21	0.13	0.01	0.06	954
Boundary Creek	8 - Blue Joe	10	0	9	0	0	0	1	0	N/A	0.11	N/A	0.89	338
Boundary Creek	9 - Saddle	30	30	0	0	0	0	0	0	0.02	0.00	0.00	0.00	960
Boundary Creek	10-Marsh Creek	22	22	0	0	0	0	0	0	0.13	0.00	0.00	0.00	702
Burton Creek	1	11	1	0	0	0	8	2	0	N/A	0.91	N/A	0.33	344

Table 6 (continued)

Stream	Site	N	O. Mykiss	O. Clarkii	F 1	F 2	Rainbow _BC	Cutthroat _BC	Other _HYB	HE	Hybridization	Intraspecific Introgression	Interspecific Introgression	Total alleles
Caboose Creek	Genetics	30	18	0	0	4	5	0	3	0.25	0.40	0.11	0.10	950
Caribou Creek	1	21	12	1	2	0	3	2	1	0.23	0.38	0.08	0.22	650
Curley Creek	Genetics	12	12	0	0	0	0	0	0	0.29	0.00	0.36	0.00	384
Dodge Creek	1	30	30	0	0	0	0	0	0	0.08	0.00	0.00	0.00	958
Dodge Creek	Lower	30	30	0	0	0	0	0	0	0.23	0.00	0.11	0.00	960
EF Boulder Creek	1	30	30	0	0	0	0	0	0	0.24	0.00	0.99	0.01	956
EF Boulder Creek	2	30	30	0	0	0	0	0	0	0.24	0.00	0.96	0.01	960
EF Boulder Creek	3	30	30	0	0	0	0	0	0	0.21	0.00	0.99	0.01	950
Fisher Creek	1	29	12	0	10	0	4	3	0	0.24	0.59	0.05	0.26	900
John Crown	1	30	0	30	0	0	0	0	0	N/A	0.00	N/A	0.01	958
John Crown	Genetics	30	30	0	0	0	0	0	0	0.29	0.00	0.43	0.00	960
Katka Creek	Genetics	30	29	0	0	0	1	0	0	0.28	0.03	0.65	0.01	946
Long Canyon Creek	1	30	23	0	2	1	3	1	0	0.22	0.23	0.02	0.09	952
Long Canyon Creek	2	30	30	0	0	0	0	0	0	0.16	0.00	0.02	0.00	912
Moyie River	Genetics	30	30	0	0	0	0	0	0	0.28	0.00	0.24	0.00	960
NF Callahan Creek	1	29	29	0	0	0	0	0	0	0.11	0.00	0.00	0.00	916
NF Callahan Creek	2	30	30	0	0	0	0	0	0	0.11	0.00	0.00	0.00	936
Parker Creek	1	19	1	4	9	0	2	2	1	N/A	0.74	N/A	0.44	580
Parker Creek	2	30	0	30	0	0	0	0	0	N/A	0.00	N/A	0.00	956

Table 6 (continued)

Stream	Site	N	O. Mykiss	O. Clarkii	F 1	F 2	Rainbow _BC	Cutthroat _BC	Other _HYB	HE	Hybridization	Intraspecific Introgression	Interspecific Introgression	Total alleles
Ruby Creek	1	30	29	0	0	0	1	0	0	0.22	0.03	0.06	0.01	958
Ruby Creek	2	30	30	0	0	0	0	0	0	0.22	0.00	0.99	0.00	960
SF Callahan Creek	1	30	30	0	0	0	0	0	0	0.16	0.00	0.02	0.00	956
SF Callahan Creek	2	29	29	0	0	0	0	0	0	0.14	0.00	0.01	0.00	924
Snow Creek	1	20	0	20	0	0	0	0	0	N/A	0.00	N/A	0.00	640
Snow Creek	2	30	0	30	0	0	0	0	0	N/A	0.00	N/A	0.00	958
Snow Creek	3	31	0	31	0	0	0	0	0	N/A	0.00	N/A	0.01	992
Trail Creek	1	29	29	0	0	0	0	0	0	0.22	0.00	0.03	0.00	928
Trail Creek	2	29	27	0	0	0	1	0	1	0.21	0.07	0.02	0.02	924
Trail Creek	Cone Creek	30	27	0	0	0	2	0	1	0.20	0.10	0.01	0.03	960
Trout Creek	1	31	0	12	5	1	1	10	2	N/A	0.61	N/A	0.22	960
Trout Creek	2	30	0	30	0	0	0	0	0	N/A	0.00	N/A	0.01	956
Trout Creek	3	30	0	28	0	0	0	0	2	N/A	0.07	N/A	0.03	960
Twentymile Creek	1	30	30	0	0	0	0	0	0	0.20	0.00	0.01	0.00	960
Twentymile Creek	2	30	30	0	0	0	0	0	0	0.19	0.00	0.01	0.00	960
Twentymile Creek	3	29	29	0	0	0	0	0	0	0.10	0.00	0.01	0.00	928

Table 7. Comparison of *Oncorhynchus mykiss* and *O. clarkii* hybridization and interspecific introgression rates from selected Kootenai River tributary sites sampled in 2018 and similar stream reaches sampled in 2006. Introgression rates are based on allele frequencies of the dominant species found at each site. Stream reaches where hybridization increased between 2006 and 2018 are denoted by an asterisk (*).

Stream	Site	O. mykiss	O. clarkii	Hybridization	Introgression	Site Code_08	Hybridization_08	Introgression_08
Ball Creek	1*	14	5	36.7%	66.6%	BAL	18.5%	4.1%
Ball Creek	2	0	31	0.0%	0.7%	BAU-A	6.0%	7.5%
Ball Creek	3	0	29	0.0%	0.2%	BAU-B	0.0%	0.0%
Boulder Creek	3	0	27	10.0%	4.7%	BOU	14.8%	1.1%
EF Boulder Creek	2	30	0	0.0%	0.6%	EFB	10.5%	0.8%
Boundary Creek	3 - Shorty	27	0	0.0%	0.0%	SH1/SHL	0.0%	0.0%
NF Callahan Creek	1	29	0	0.0%	0.0%	NFA-M	0.0%	0.0%
NF Callahan Creek	2	30	0	0.0%	0.0%	NFA-U	0.0%	0.0%
SF Callahan Creek	2	29	0	0.0%	0.0%	SFA	6.7%	0.5%
SF Callahan Creek	1	30	0	0.0%	0.0%	SFA	6.7%	0.5%
Long Canyon Creek	2	30	0	0.0%	0.0%	LON	0.0%	0.0%
Ruby Creek	2	30	0	0.0%	0.0%	RU3	0.0%	0.0%
Snow Creek	2	0	30	0.0%	0.1%	SNM	3.3%	0.3%
Snow Creek	3	0	31	0.0%	1.0%	SNU-A	0.0%	0.0%
Trout Creek	2	0	30	0.0%	0.5%	TCU	0.0%	0.0%
Twentymile Creek	2	30	0	0.0%	0.4%	TWE	10.8%	0.8%

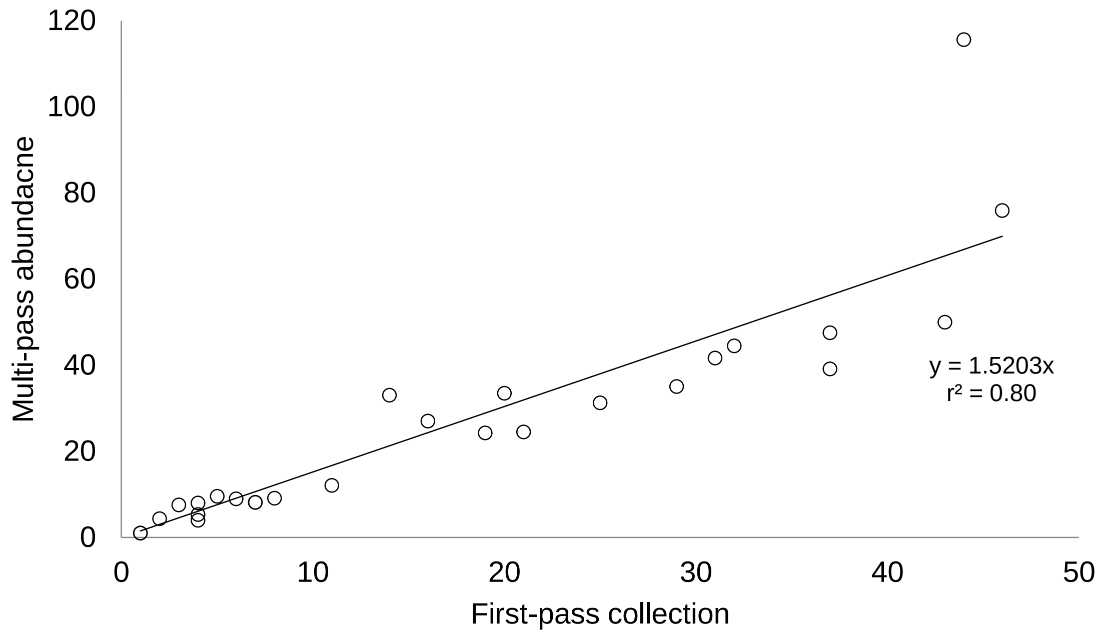

Figure 1. Linear model of backpack electrofishing multi-pass depletion estimates by first-pass catch from Idaho tributaries of the Kootenai River sampled in 2018.

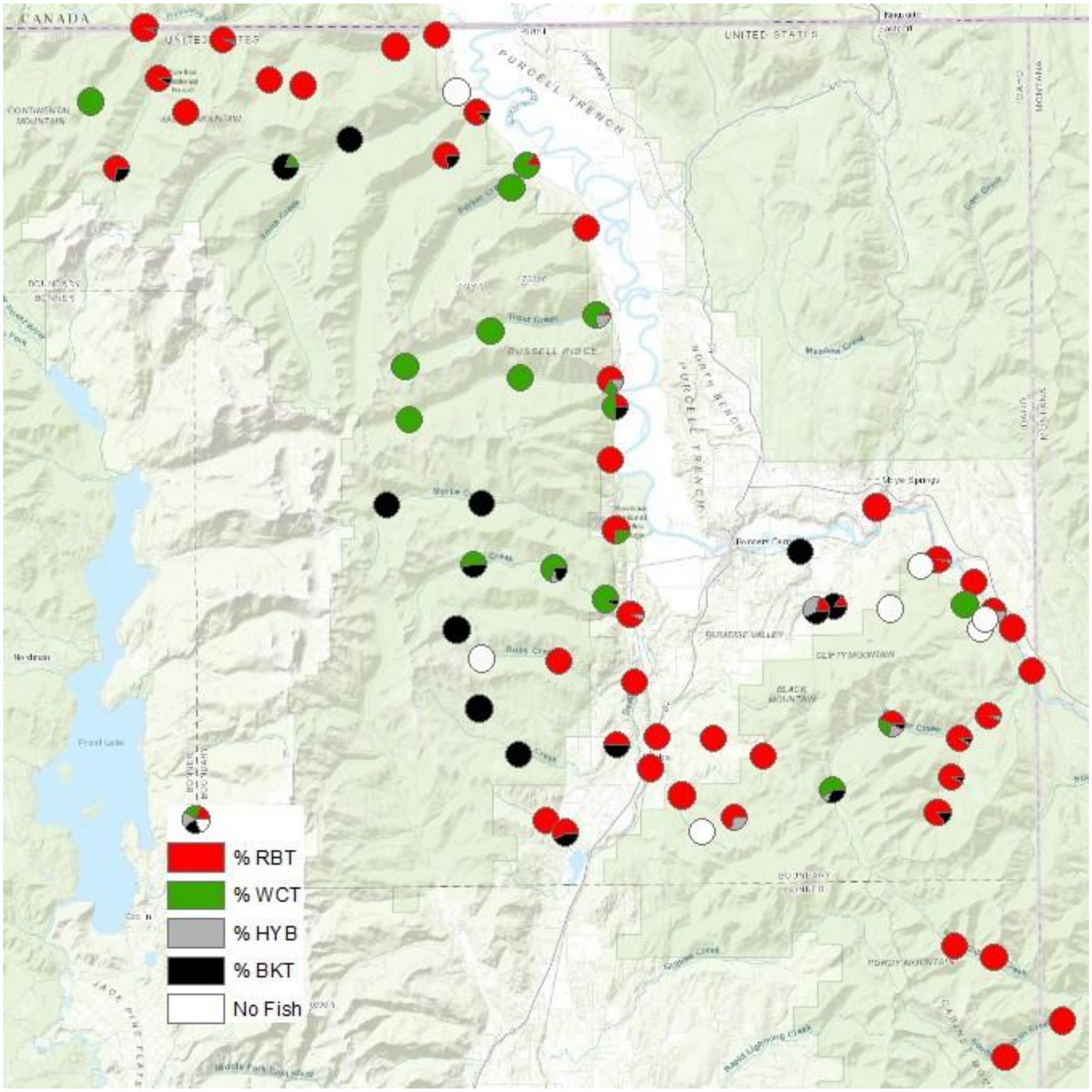

Figure 2. Proportional catch of phenotypically identified Rainbow Trout, Westslope Cutthroat Trout, Rainbow x Westslope Cutthroat trout hybrids, and Brook Trout from Idaho tributaries to the Kootenai River sampled in 2018.

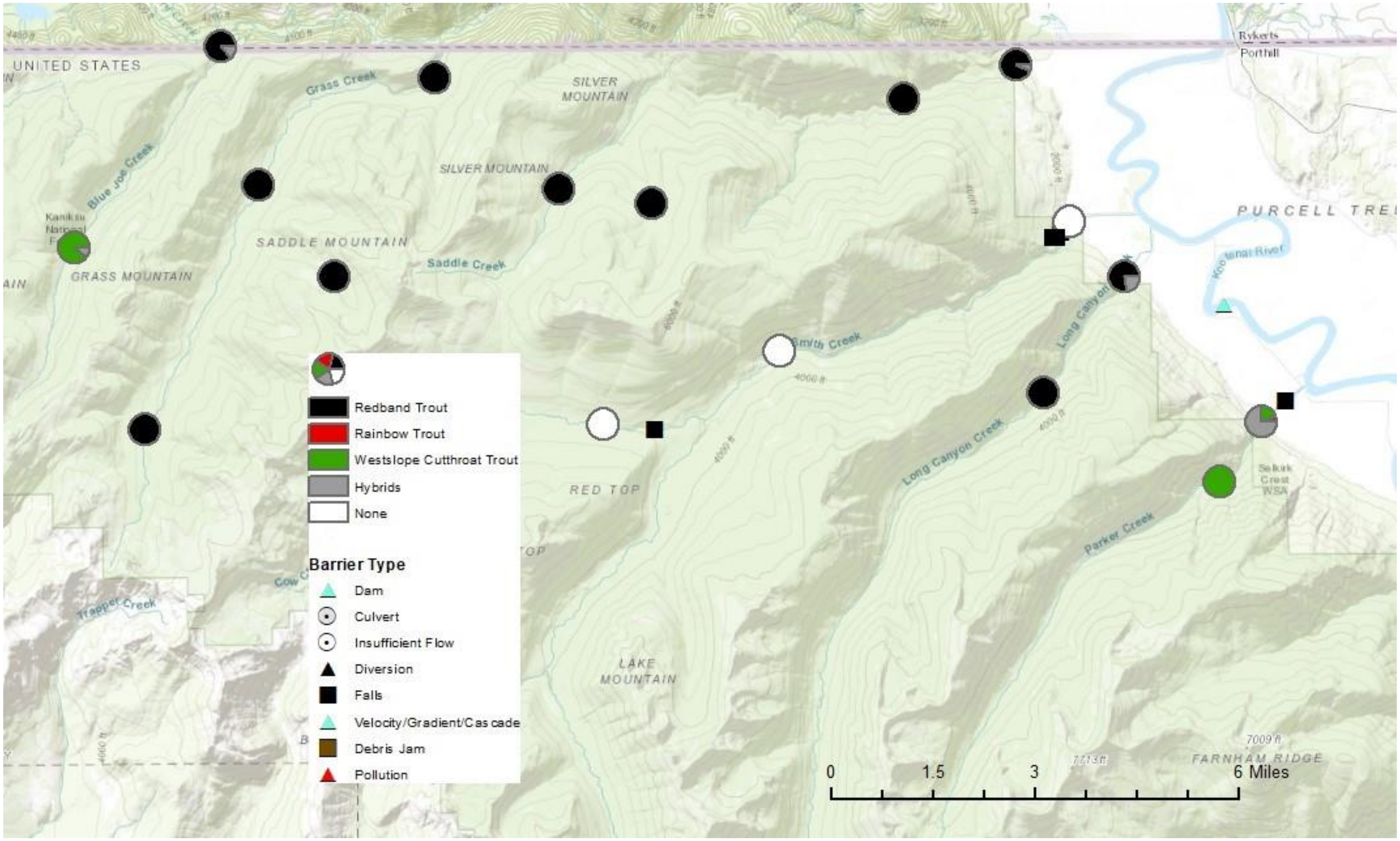

Figure 3. Proportional catch of genotypically-identified Oncorhynchus spp. from Boundary Creek, Smith Creek, Long Canyon Creek, and Parker Creek, Idaho. All are tributaries to the Kootenai River that were sampled in 2018. Redband represent O. mykiss samples with low (< 10%) intraspecific introgression and low (<10%) interspecific introgression. Barriers to fish movement are depicted in tributary locations where they are known to occur.

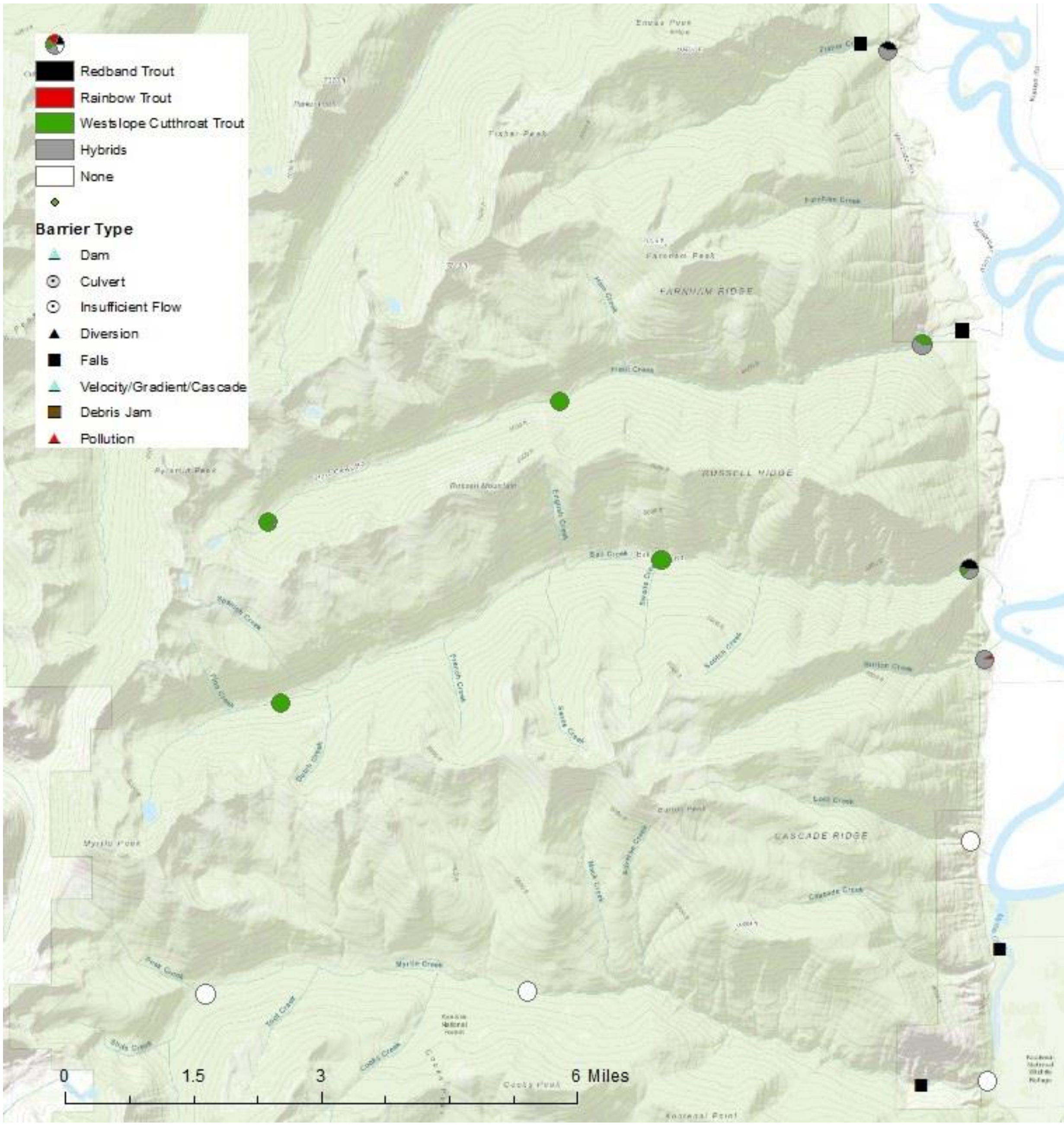

Figure 4. Proportional catch of genotypically identified Oncorhynchus spp. from Fisher Creek, Trout Creek, Ball Creek, Burton Creek, Lost Creek, and Myrtle Creek, Idaho. All are tributaries to the Kootenai River that were sampled in 2018. Redband represent O. mykiss samples with low (< 10%) intraspecific introgression and low (<10%) interspecific introgression. Barriers to fish movement are depicted in tributary locations where they are known to occur.

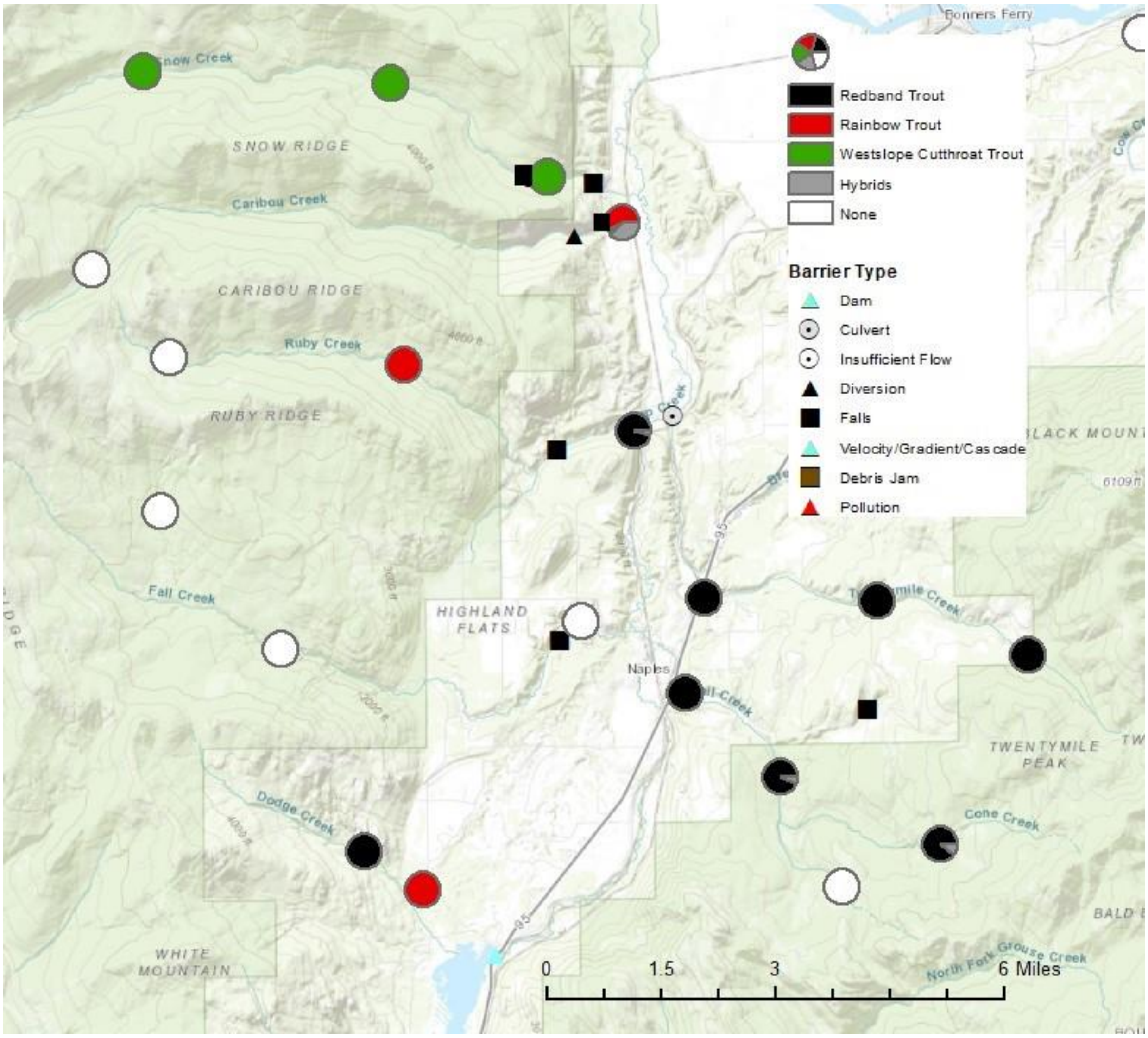

Figure 5. Proportional catch of genotypically identified Oncorhynchus spp. from Snow Creek, Caribou Creek, Ruby Creek, and Fall Creek, Dodge Creek, Trail Creek, and Twentymile Creek, Idaho. All are tributaries to the Kootenai River sampled in 2018. Redband represent O. mykiss samples with low (< 10%) intraspecific introgression and low (<10%) interspecific introgression. Barriers to fish movement are depicted in tributary locations where they are known to occur.

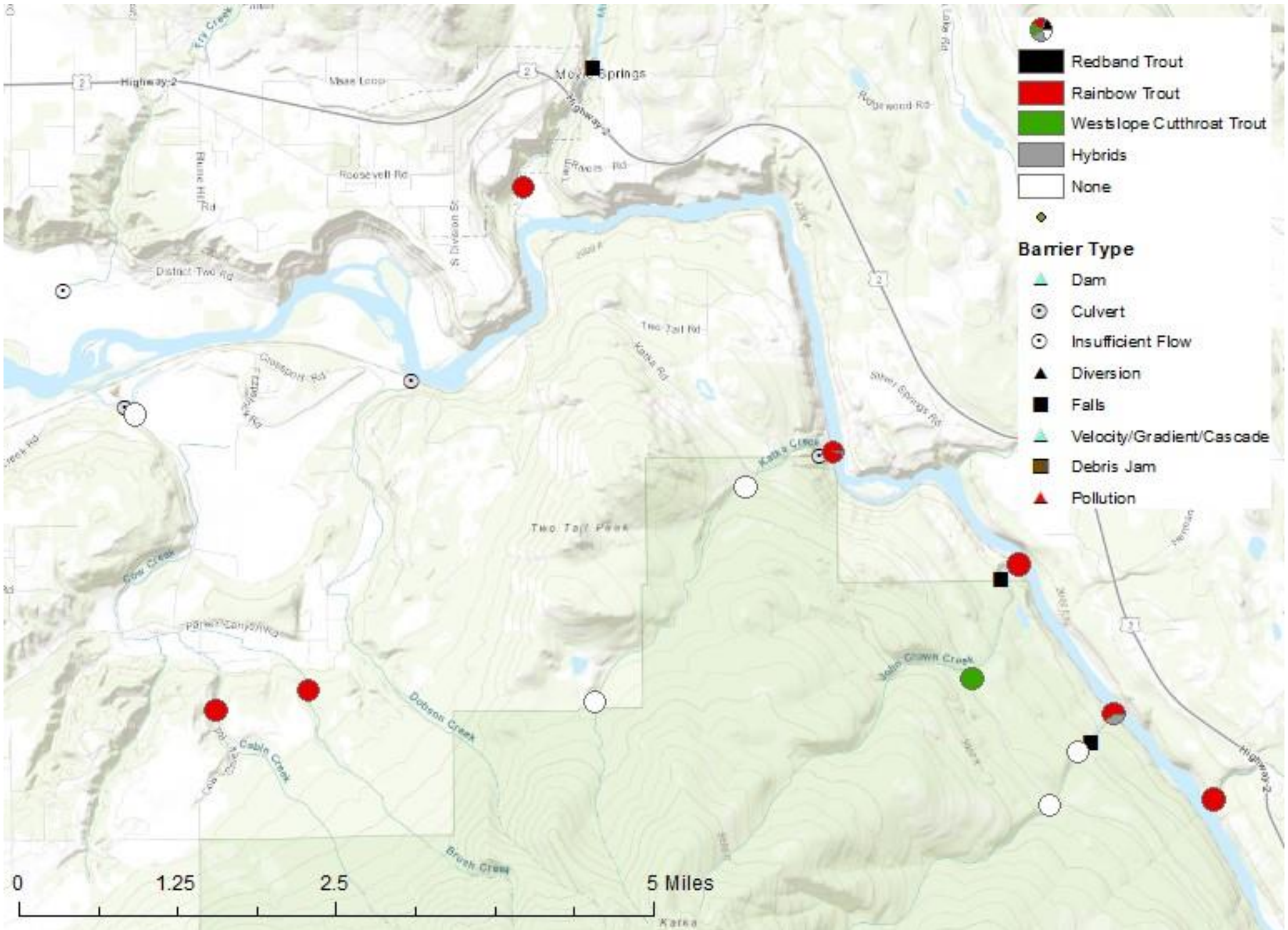

Figure 6. Proportional catch of genotypically identified Oncorhynchus spp. from Cow Creek, Moyie River, Katka Creek, John Crown Creek, Caboose Creek, and Curley Creek, Idaho. All are tributaries to the Kootenai River sampled in 2018. Redband represent O. mykiss samples with low (<10%) intraspecific introgression and low (<10%) interspecific introgression. Barriers to fish movement are depicted in tributary locations where they are known to occur.

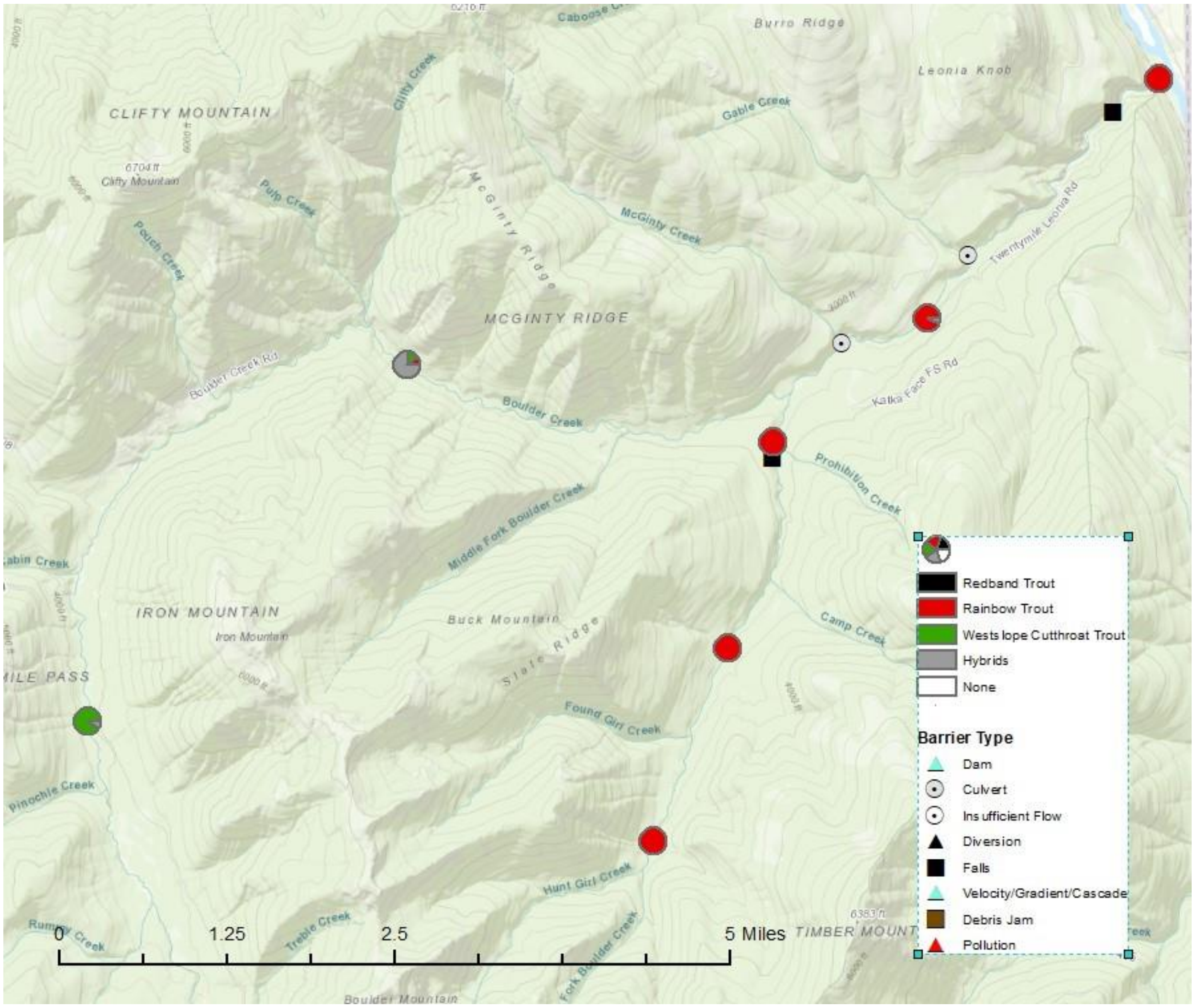

Figure 7. Proportional catch of genotypically identified Oncorhynchus spp. from Boulder Creek, and East Fork Boulder Creek, Idaho. All are tributaries to the Kootenai River sampled in 2018. Redband represent O. mykiss samples with low (< 10%) intraspecific introgression and low (<10%) interspecific introgression. Barriers to fish movement are depicted in tributary locations where they are known to occur.

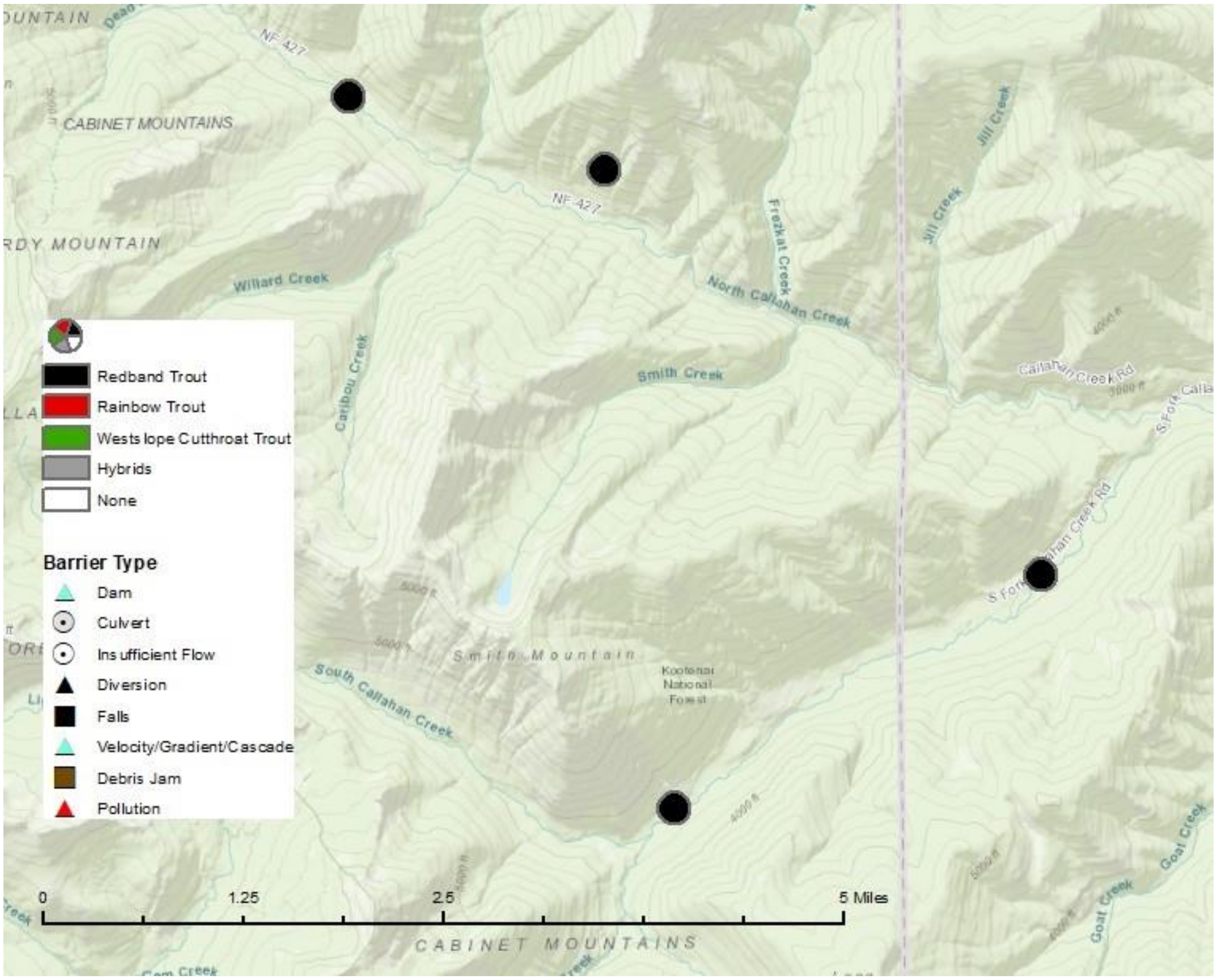

Figure 8. Proportional catch of genotypically identified Oncorhynchus spp. from North Fork Callahan Creek and South Fork Callahan Creek, Idaho. All are tributaries to the Kootenai River sampled in 2018. Redband represent O. mykiss samples with low (< 10%) intraspecific introgression and low (<10%) interspecific introgression. Barriers to fish movement are depicted in tributary locations where they are known to occur.

Figure 9. NJ-tree based on FST for all O. mykiss genetic samples collected in Idaho tributaries to the Kootenai River in 2018. Figure includes reference hatchery Rainbow Trout (i.e., coastal origin).

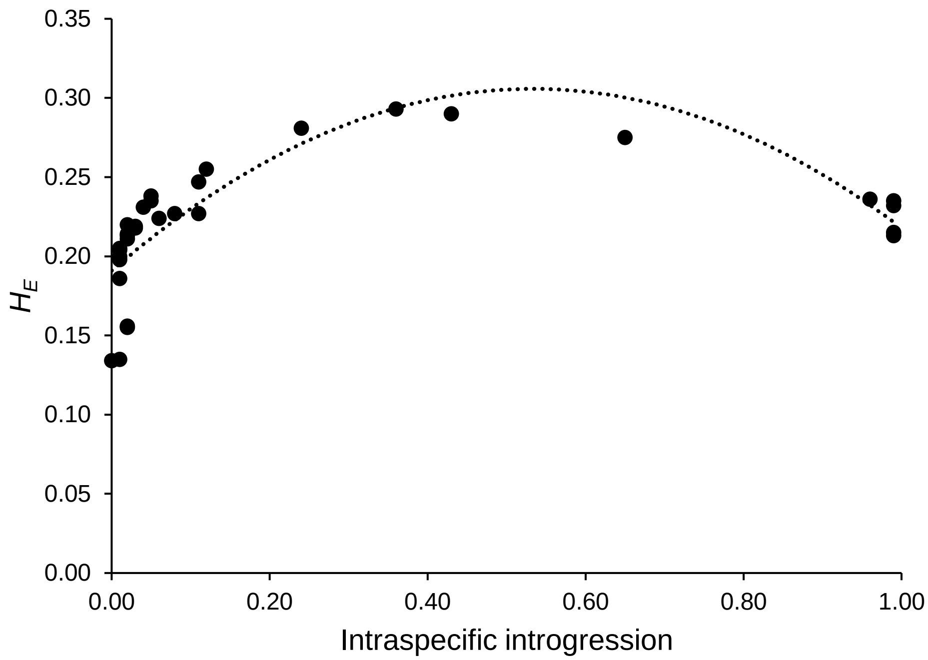

Figure 10. Expected heterozygosity (HE) of O. mykiss over a range of intraspecific introgression in genetic sample from Idaho tributaries to the Kootenai River.

Appendix A. Identified opportunities to improve or restore native fish populations in tributaries to the Kootenai River, Idaho.

Stream	Target Species	Action
Ruby Creek	Redband Trout	Remove Rainbow Trout (coastal origin) in the upper drainage by rotenone treatment. Leave fishless or introduce Redband Trout
Cow Creek	Redband Trout	Remove existing culvert barrier low in the drainage. Improve degraded habitat. Remove mixed species fish community by rotenone treatment. Restore (or) introduce Redband Trout.
Myrtle Creek	Westslope Cutthroat Trout	Remove Brook Trout from the majority of the drainage by rotenone treatment. Restore (or) introduce Westslope Cutthroat Trout
Fall Creek	Westslope Cutthroat Trout	Remove Brook Trout from the majority of the drainage by rotenone treatment. Restore (or) introduce Westslope Cutthroat Trout
Boulder Creek	Westslope Cutthroat Trout	Remove Rainbow Trout (coastal origin) in the upper drainage by rotenone treatment. Restore (or) introduce Westslope Cutthroat Trout
East Fork Boulder Creek	Westslope Cutthroat Trout	Remove Rainbow Trout (coastal origin) in the upper drainage by rotenone treatment. Restore (or) introduce Westslope Cutthroat Trout

UPPER PRIEST LAKE LAKE TROUT MANAGEMENT

ABSTRACT

Upper Priest Lake is currently managed for the conservation of native species. In support of this objective, removal of non-native Lake Trout has occurred since 1998. In 2018, gill nets were used to remove 2,425 Lake Trout during a two-week period from May 14 to May 25. Average daily catch rate from standard mesh sizes was 14.3 fish/box (± 2.8, 80% C.I.), which was similar to recent years. Lake Trout length ranged from 90 to 950 mm. Bull trout catch rate (0.07 fish/box) was below average when compared to the previous ten-year period. Trend data suggest that Lake Trout abundance remained stable and low, supporting continuation of removal efforts to benefit native fishes in Upper Priest Lake.

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INTRODUCTION

Native fishes, including Bull Trout *Salvelinus confluentus* and Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi,* played an important role in the history of Priest and Upper Priest lake fishing. Historically, Bull Trout provided a harvest-oriented trophy fishery in Priest and Upper Priest lakes (Bjornn 1957). However, harvest opportunities were discontinued in 1984 following declines in Bull Trout abundance. Although the influence of fishing mortality on the population was removed, a positive population response did not occur (Mauser et al. 1988). Today, the Bull Trout population in Upper Priest Lake is considered depressed while the population in Priest Lake is considered functionally lost (DuPont et al. 2007). Native Westslope Cutthroat Trout were also historically abundant in Priest and Upper Priest lakes and provided the primary fishery in both lakes prior to the 1950s (Mauser et al. 1988). Westslope Cutthroat Trout harvest opportunities were closed in 1988, following a perceived decline in overall abundance. Overharvest, interspecific competition, predation, and degradation of spawning habitat were all believed to contribute to the decline of native fish in this system.

Although multiple factors have likely influenced the abundance of native fishes in Priest and Upper Priest lakes, increasing Lake Trout *Salvelinus namaycush* abundance was the primary cause of population-scale changes in native fish communities. Lake Trout, where introduced as a non-native sport fish, are often linked to negative responses in other native and non-native species through predation and/or competition (Martinez et al. 2009). In Upper Priest Lake, Lake Trout were not known to be abundant until the late 1990s (Fredericks 1999). By 1998, Lake Trout abundance in Upper Priest Lake was estimated to be 859 fish (Fredericks 1999). At that time, fishery managers were concerned native fish communities in Upper Priest Lake were at risk.

Native fish conservation has been an ongoing management focus on Upper Priest Lake. In an effort to reduce the potential impacts of Lake Trout on native fish populations in Upper Priest Lake, the Idaho Department of Fish and Game (IDFG) began a Lake Trout removal program in 1998. Gill nets have been used annually to remove Lake Trout and reduce their abundance in the lake. These management efforts have removed between 150 and 5,000 Lake Trout annually from Upper Priest Lake (Fredericks et al. 2013). In 2018, we continued Lake Trout reduction efforts in Upper Priest Lake with the intent of benefiting native fish species.

OBJECTIVE

Conserve native fish populations in Upper Priest Lake by maintaining low Lake Trout abundance.

STUDY SITE

Upper Priest Lake is located approximately 21 km south of the Idaho-British Columbia border in the northwest corner of the Idaho Panhandle. It is a glacial lake that has roughly 13 km of shoreline, a surface area of 566 hectares (ha), a maximum depth of approximately 31 meters (m) and a maximum surface temperature of approximately 21 ˚C. The lake is bathtub-shaped with steep shoreline slopes and a flat bottom. Upper Priest and Priest lakes are held at 743 m elevation from the end of spring runoff until mid-October, which is controlled by a low-head dam located at the outlet of Priest Lake. Upper Priest Lake is connected to Priest Lake by a channel known as the Thorofare. The Thorofare is roughly 3.2 km long, 70 m wide and 1.5-3 m deep at summer pool. At low pool, water depth in the Thorofare outlet is < 0.15 m and prohibits most boat traffic.

METHODS

We completed the 2018 Upper Priest Lake Lake Trout removal effort between May 14 and May 25. Hickey Brothers Research, LLC was contracted to provide equipment and labor for completion of the netting project. An 11-m commercial gill net boat was used to complete sampling efforts. Funding for completion of the Lake Trout removal effort was provided by the United States Fish and Wildlife Service (USFWS), Kalispel Tribe, and Idaho Department of Fish and Game.

We used monofilament sinking gill nets to capture and remove Lake Trout from Upper Priest Lake. Individual gill net dimensions were 91 m long by 2.7 m high. Multiple nets were tied together end-to-end to create a single net gang. Collectively, the net gang was comprised of a range of mesh sizes. Standardized mesh sizes (stretch-measure) were 45, 51, 64, 76, 89, 102, 114, and 127 mm (Table 8). Fishing effort was measured in units defined as net boxes. Boxes were used to transport nets onboard the boat, and each box of net was equivalent to approximately 273 m or three 91-m nets. Daily effort was split between morning and afternoon sets each day. The combined effort per day was 30 boxes of gill net. A total of 240 boxes of gill net were placed over ten days. Both morning and afternoon sets were made on each day, except the first and last days of each work week during which only one set was made on each date. The combined total effort for the first and last day of each work week was 30 boxes of net. Typically 18 boxes of net were set in the morning and 12 boxes of net were set in the afternoon. The combined effort by mesh size was consistent within morning and afternoon sets, respectively. The time between net placement and initiating net lifting varied from two to five hours for all sets. Gill net was set throughout Upper Priest Lake over the course of the sampling period at depths varying from 10 to 31 m. Placement of nets in and around the primary inlets and outlet of Upper Priest Lake was avoided to reduce bycatch of Bull Trout and Westslope Cutthroat Trout.

Relative abundance of Lake Trout in Upper Priest Lake was measured as average daily catch per unit of effort (CPUE) or fish per net box per day for catch associated with 51, 64, and 76 mm gill net mesh sizes. These mesh sizes were selected as standards because they represented the longest time series of mesh sizes fished during Upper Priest Lake removal efforts. We compared these standardized catch rates to prior years to evaluate trends in abundance. We only used data from 2010 to 2018 because catch by mesh was not recorded prior to 2010. We calculated 80% confidence bounds around estimates of average daily catch rate and used those bounds to infer differences in catch rate between years. We also evaluated change in size structure of the Lake Trout catch using catch rate from individual gill net mesh sizes. Lake Trout length was found to generally increase with gill net mesh size (Ryan et al. 2014) suggesting mesh-specific catch rates provide a relative measure of size-specific abundance. We compared mesh-specific catch rates from 2014 and 2018. Prior to 2014, a standard set of mesh sizes was not used and limited complete comparisons with prior years.

All Lake Trout caught during netting efforts were measured for total length (mm) and examined for marks. A portion of the Lake Trout catch greater than 400 mm were cleaned, packed on ice, and distributed to local food banks. Remaining Lake Trout were dispatched and returned to the lake.

Bycatch of non-target species associated with the removal effort was generally noted and fish were released if alive, though not all individuals were recorded. However, total length and condition were collected from all Bull Trout. Bull Trout condition was ranked from zero to three, with zero representing mortality and three representing excellent condition. We reported Bull Trout catch rate as the average of daily catch per unit of effort or fish per net box per day among all mesh sizes and compared catch rates from 2007 to 2018. Variance around catch rate estimates was described using 80% confidence bounds. Confidence bounds were only estimated for years during which standardized gill net effort and mesh were used (i.e., 2014-2017). A PIT tag was inserted into the dorsal sinus of each live-released Bull Trout. Future recaptures will be used to generally describe recapture rates and survival of Bull Trout encountered in netting efforts over time.

RESULTS

We removed 2,425 Lake Trout during the ten-day gillnetting effort. Average daily catch rate from 51-, 64-, and 76-mm mesh sizes was 14.3 fish/box (± 2.8, 80% C.I.; Figure 11) and demonstrated a long-term negative trend. Mesh-specific catch rates differed from those observed in 2017. Increased catch rates in 45- and 51-mm mesh sizes represented the most dramatic changes observed in 2018 (Figure 12).

Total lengths of Lake Trout varied from 90 to 950 mm (Table 8, Figure 13). In general, fish length increased with increased gill net mesh size. Small mesh sizes (45, 51, and 64 mm) had the highest catch rates and accounted for 79% of the total catch. These mesh sizes also represented 60% of total effort expended.

Incidentally caught species included Bull Trout, kokanee *Oncorhynchus nerka*, Longnose Sucker *Catostomus catostomus*, Largescale Sucker *C. macrocheilus*, Northern Pikeminnow *Ptychocheilus oregonensis*, and Peamouth *Mylocheilus caurinus*. We caught 17 Bull Trout, representing an average daily catch rate of 0.07 Bull Trout per box of net. This catch rate was below the average rate observed over the previous ten years (0.16 Bull Trout per box, Figure 14). Bull Trout caught varied from 201 to 635 mm and averaged 367 mm. The majority of Bull Trout caught in gill nets were in good to fair condition upon capture. These fish were PIT tagged and released. Direct mortality of bycaught Bull Trout in gill nets was 24%.

DISCUSSION

Collectively, catch rates suggested Lake Trout abundance remained low. Evidence exists to suggest native fishes have benefited from maintaining reduced Lake Trout abundance in Upper Priest Lake. For example, Bull Trout redd counts in Upper Priest Lake tributaries in 2017 were higher than ever previously observed and demonstrated a positive long term trend (Ryan et al., 2020c). This example not only suggested ongoing Lake Trout reductions are beneficial to Bull Trout, but that bycatch-related mortality associated with the use of gill nets in this project is also inconsequential relative to the benefits. Although evidence suggests native fish populations have benefited, Bull Trout catch rate in our netting effort was low relative to catch rates in some previous years. This inconsistency highlights a need to cautiously interpret Bull Trout catch rates resulting from this spring gill netting effort. A number of environmental variables may influence Bull Trout catch in gill nets during this period. In addition, gill nets set during the Lake Trout removal effort are specifically avoided in some areas of Upper Priest Lake with the intent of minimizing Bull Trout bycatch.

Mesh-specific catch rates provided insight into fine-scale changes in the Upper Priest Lake Lake Trout population. Specifically, catch rates reflected and increase in relative abundance of small Lake Trout from 2017. Catch rates within small mesh sizes had been stable since 2015. However, we observed average catch rates in 45- and 51-mm gill net mesh sizes increased above rates observed since that time. A trend in size structure was not evident in prior annual removal efforts to suggest a strong size class of fish was present or expected to grow into a vulnerable size range for 45- or 51-mm gill net mesh sizes (Watkins et al., 2019, Ryan et al., 2020a, Ryan et al., 2020b, Ryan et al. 2020c). However, gill net mesh sizes used in our effort may not efficiently capture the smallest Lake Trout in the population, making it difficult to predict their presence prior to recruiting to our gear. Immigration of Lake Trout from Priest Lake also remains a possible source of Lake Trout for Upper Priest Lake. Movement of Lake Trout between Priest Lake and Upper Priest Lake is known to occur (Fredericks and Venard 2001) and has been assumed to be a factor influencing Lake Trout abundance in Upper Priest Lake.

Lake Trout presence in Upper Priest Lake is the primary concern relative to the conservation of native species. Currently, catch rates suggest the Lake Trout population in Upper Priest Lake remains low and suppression efforts are successfully preventing population growth. Therefore, the negative impacts that Lake Trout pose to native species are being minimized. As such, we recommend continuation of Lake Trout removal efforts in Upper Priest Lake as a tool for conserving native fishes.

RECOMMENDATIONS

1. Continue annual gillnetting at existing levels on Upper Priest Lake to conserve native fishes.

Table 8. Gill net effort and Lake Trout (LKT) catch by gill net mesh size in Upper Priest Lake, Idaho during 2018. Total length (TL) ranges of Lake Trout caught are reported for individual gill net mesh sizes.

Mesh	Effort (m)	% of Total effort	LKT caught	LKT/box	Min TL	Max TL
45 mm	13,167	20%	552	11.5	90	764
51 mm	13,167	20%	867	18.1	189	705
64 mm	13,167	20%	498	10.4	163	765
76 mm	4,389	7%	128	8.0	315	883
89 mm	4,389	7%	107	6.7	301	797
102 mm	8,778	13%	179	5.6	327	806
114 mm	4,389	7%	49	3.1	185	895
127 mm	4,389	7%	45	2.8	205	950

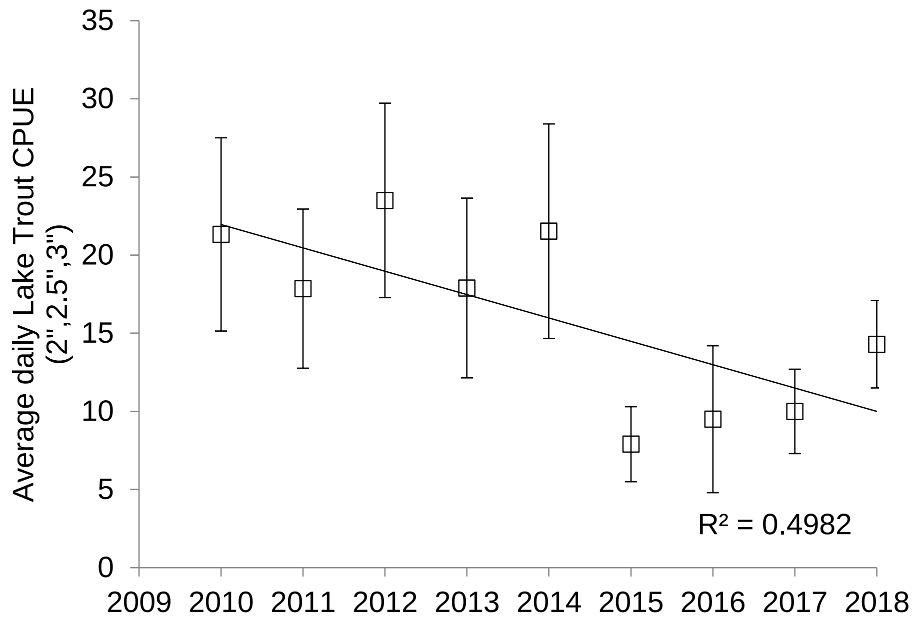

Figure 11. Average daily Lake Trout catch rates and 80% confidence intervals by year from combined standard gill net mesh sizes (51 mm, 64 mm, and 76 mm) fished in Upper Priest Lake, Idaho from 2010 through 2018.

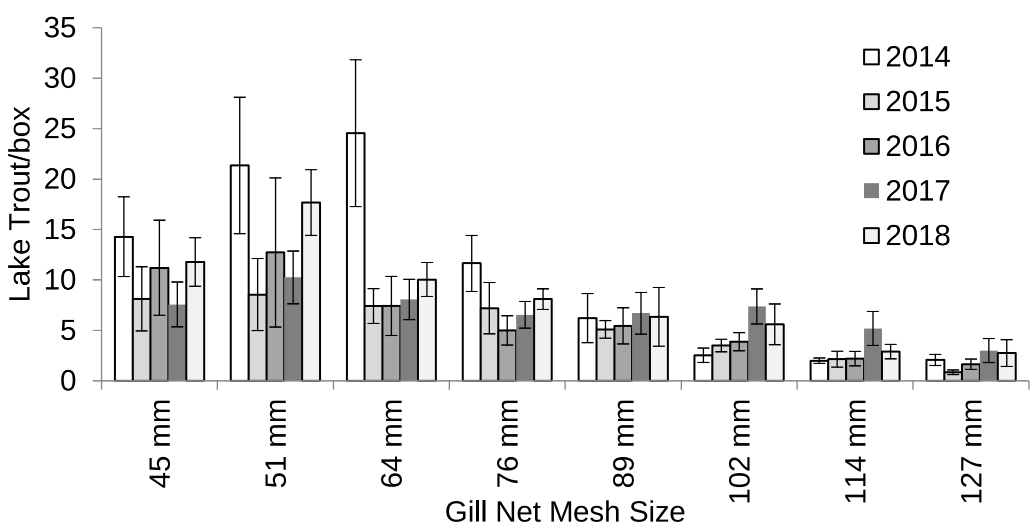

Figure 12. Average daily Lake Trout catch rate (Lake Trout/box) and 80% confidence intervals by mesh size from all standardized gill nets fished in Upper Priest Lake, Idaho from 2014 through 2018.

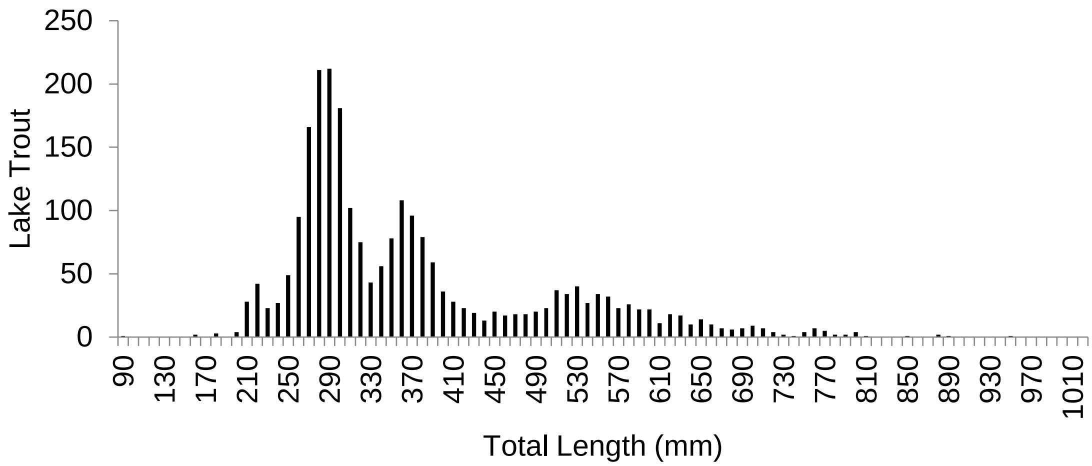

Figure 13. Length-frequency distribution of Lake Trout sampled in Upper Priest Lake, Idaho during 2018.

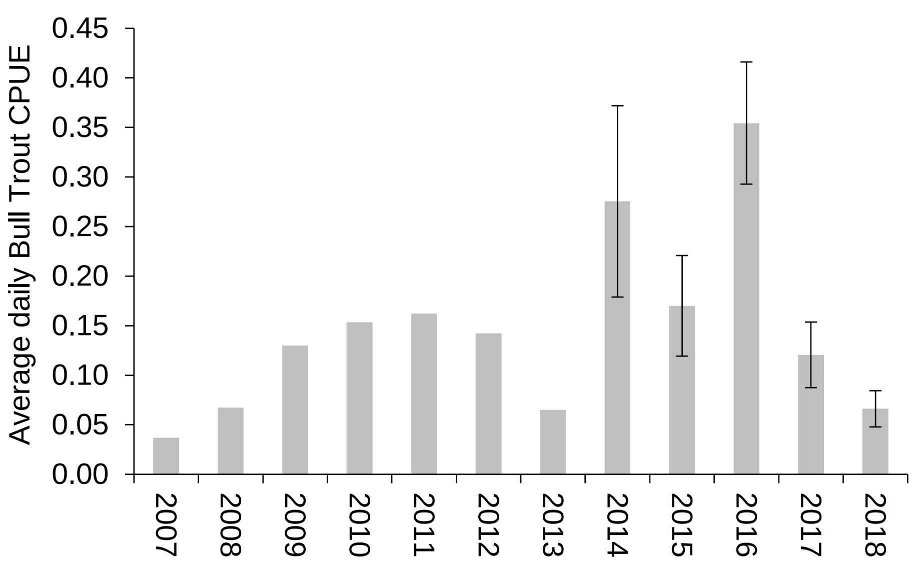

Figure 14. Average daily Bull Trout catch rate (Bull Trout/box) and 80% confidence intervals from all mesh sizes fished in Upper Priest Lake, Idaho from 2007 through 2018. Confidence intervals were only estimated for years in which gill nets mesh and effort were standardized.

PRIEST LAKE FISHERY INVESTIGATIONS

ABSTRACT

In 2018, we investigated Priest Lake kokanee *Oncorhynchus nerka* abundance in an effort to describe population trends. We conducted a lakewide mobile acoustic survey to estimate kokanee abundance. We also monitored kokanee spawner abundance in Priest Lake by counting mature spawning adults at five standard areas. Estimated density of Priest Lake kokanee in August 2018 was 21 fry/ha and 11 age-1 to age-4 kokanee/ha. A total of 4,395 kokanee adults were observed along five shoreline areas of Priest Lake in November. The combined observations from surveys suggest kokanee densities remain low.

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INTRODUCTION

Priest Lake is located in Idaho's Panhandle Region approximately 28 km south of the Canadian border. Surface area of the lake is 9,446 ha with 8,190 ha of pelagic habitat greater than 12 m deep. Historically, Priest Lake provided fisheries for Bull Trout *Salvelinus confluentus*, Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi*, and Mountain Whitefish *Prosopium williamsoni*. Introductions of kokanee *Oncorhynchus nerka*, Lake Trout *Salvelinus namaycush,* Largemouth Bass *Micropterus salmoides,* Smallmouth Bass *Micropterus dolomieu*, and Yellow Perch *Perca flavescens* created additional fishing opportunities that are present today (Watkins et al. 2018).

Priest Lake fisheries management has changed significantly since the early 1900s. Bull Trout were once a primary target of anglers, but have been regulated under a "no harvest" scenario since the late 1980s due to declines in abundance. Similarly, perceived declines in Westslope Cutthroat Trout have also led to a catch and release only fishery. Kokanee also once offered the primary fishery in the lake and a significant harvest opportunity. However, kokanee abundance declined through the 1970s and 80s resulting in fishery closure. Kokanee densities in the lake remain low, but a harvest fishery was re-established in 2011 and has gained considerable interest among anglers (Fredericks et al. 2013). Lake Trout, once less common in the catch, provided a trophy opportunity prior to kokanee collapse. However, increased Lake Trout abundance between the 1970s and 1990s led to shifting management objectives and the current yield fishery (IDFG 2013). Recently, Smallmouth Bass were unintentionally established in Priest Lake and have gained angler interest. Mysid shrimp *Mysis diluviana* were introduced to Priest Lake in the 1960s and are assumed to have positively influenced Lake Trout and negatively influenced other once–abundant fish species (i.e., kokanee, Bull Trout, Westslope Cutthroat Trout; IDFG 2013).

Current management of the Priest Lake fishery is focused on providing a mix of angling opportunities, primarily for Lake Trout, kokanee, and Westslope Cutthroat Trout. In 2018, we conducted surveys of kokanee abundance to describe current population trends and the opportunity kokanee provide to anglers.

METHODS

Acoustic Kokanee Survey

We conducted a lakewide mobile acoustic survey on Priest Lake to estimate kokanee abundance on the night of August 6, 2018. We used a Simrad EK60 split-beam, scientific echosounder with a 120 kHz transducer to estimate kokanee abundance. Ping rate was set at 0.3 to 0.5 seconds per ping. A pole-mounted transducer was located 0.66 m below the surface, off the port side of the boat, and pointed downward. The echosounder was calibrated prior to the survey using a 23 mm copper calibration sphere to set the gain and to adjust for signal attenuation to the sides of the acoustic axis. Prior to our survey, we measured one temperature profile as a calibration of signal speed and as a reference of the expected zone of occupancy for kokanee. Water temperatures were measured at one meter intervals for 15 meters using a YSI 85-50 dissolved oxygen temperature meter (YSI Incorporated). Mean water temperature for water depths between zero and ten meters was used in system calibration. We used Simrad ER60 software (Simrad Yachting) to determine and input the calibration settings.

We used standardized transects to complete the survey (Maiolie et al. 2013). We followed a uniformly spaced, zigzag pattern of 15 transects stretching from shoreline to shoreline. The zigzag pattern was used to maximize the number of transects that could be completed in one night. The pattern followed the general rule of using a triangular design (zigzags) when the transect length was less than twice the transect spacing (Simmonds and MacLennan 2005). The starting point of the first transect at the northern end of the lake was originally chosen at random. Boat speed was approximately 2.4 m/s.

Kokanee abundance was determined using echo integration techniques. Echoview software version 8 (Echoview Software Pty Ltd) was used to view and analyze the collected data. A box was drawn around the kokanee layer on each of the echograms and integrated to obtain the nautical area scattering coefficient (NASC) and analyzed to obtain the mean target strength of all returned echoes. This integration accounted for fish that were too close together to detect as a single target (MacLennan and Simmonds 1992). Densities were then calculated by the equation:

$$\text{Density (fish/ha)} = (\text{NASC} / 4\pi 10^{TS/10})\ 0.00292$$

where: NASC is the total backscattering in m^2/nautical mile^2 and TS is the mean target strength in dB for the area sampled.

Kokanee density was estimated directly from the echograms. All fish in the observed pelagic fish layer were identified as Kokanee if target strengths of the observed fish were within the expected size range. Size ranges were based on Love's equation, which describes a relationship between target strength and length (Love 1971). A total kokanee density for all fish was calculated by echo integration. Next, a virtual echogram was built of the corrected target strengths. We then multiplied the total kokanee density estimate on each transect by the percentage of small targets (-60 dB and -45 dB) to estimate the density of kokanee fry. The percentage of large targets (-44 dB to -30 dB) were used to estimate density of kokanee age classes one to four.

We calculated kokanee abundance by multiplying estimated densities by the area of usable pelagic habitat in Priest Lake. Priest Lake has been estimated to contain 8,190 ha of pelagic habitat usable by kokanee (Maiolie et al. 2013). Eighty percent confidence intervals were calculated for the estimates of fry and older age classes of kokanee. Error bounds calculated for arithmetic mean densities utilized a Student's T distribution. The entire lake was considered to be one section, without stratification by area.

Shoreline Kokanee Count

We monitored kokanee spawner abundance in Priest Lake on November 6, 2018. Spawning kokanee were observed and counted at five standard nearshore areas, including Copper Bay, Hunt Creek, Cavanaugh Bay, Indian Creek, and Huckleberry Bay. We collected a sample of spawning kokanee adjacent to the mouth of Hunt Creek using a monofilament gill net to describe size by sex. One gillnet was set for 15 minutes. The monofilament gillnet was 46 m long with variable mesh panels from 1.9- to 6.4-mm bar mesh. Sexes were determined by examining external characteristics of each individual.

RESULTS

Acoustic Kokanee Survey

Estimated density of Priest Lake kokanee in August 2018 was 21 fry/ha (± 5.3, 80% C.I.; Table 9; Figure 15) and 11 age-1 to age-4 kokanee/ha (± 3.3, Table 9; Figure 15). Expanding these densities generated total lakewide estimates of 174,648 kokanee fry and 89,399 kokanee age-1 to 4. The thermocline was approximately at 9 m (Figure 16).

Shoreline Kokanee Count

We counted a total of 4,395 kokanee along five shoreline areas of Priest Lake in 2018 (Table 10; Figure 17). Our count was greater than observed in 2017 (Figure 17). Spawning adult kokanee collected near Hunt Creek ranged in length from 330 to 460 mm and averaged 397 mm (n = 12) and 394 mm (n = 9), for males and females, respectively.

DISCUSSION

Priest Lake kokanee abundance and other metrics described in our surveys continued to reflect a low-density kokanee population. Our acoustic estimate of age-1 to age-4 abundance represented an increase from the prior year, but was within the observed variability of recent estimates and limited our ability to conclude abundance changed significantly (Ryan et al. 2020c). Priest Lake shoreline kokanee counts also increased relative to counts in 2017, but remained low (Ryan et al. 2020c). Average length of kokanee spawners declined, presumably in response to increasing abundance, a typical pattern observed over the time series of shoreline counts.

RECOMMENDATIONS

1. Continue utilizing acoustic surveys and shoreline spawner counts as tools for monitoring Priest Lake kokanee abundance in low-density conditions.

Table 9. Acoustic kokanee survey results from Priest Lake, Idaho on August 6, 2018.

Transect	Single targets	NASC	Mean TS	Total density (fish/ha)	% fry	Fry density	% ages 1-4	Age 1-4 density
1	5	0.7	-55.5	61	1.00	61	0.00	0
2	11	5.2	-44.2	32	0.82	26	0.18	6
3	10	10.1	-38.4	16	0.70	11	0.30	5
4	8	2.5	-48.5	42	0.75	31	0.25	10
5	15	5.9	-44.9	43	0.87	37	0.13	6
6	13	2.6	-47.1	31	0.00	0	0.00	0
7	20	17.9	-39.2	34	0.60	21	0.40	14
8	27	21.9	-40.3	55	0.43	24	0.57	31
9	16	30.4	-37.7	42	0.56	23	0.44	19
10	23	15.0	-40.5	39	0.31	12	0.69	27
11	26	20.3	-39.5	42	0.61	25	0.39	16
12	21	47.8	-36.1	45	0.69	31	0.31	14
13	5	13.4	-38.3	21	0.24	5	0.76	16
14	2	0.14	-49.4	3	0.00	0	0.00	0
15	7	10.3	-37.0	12	1.00	12	0.00	0
Mean				34		21		11

Table 10. Kokanee spawner counts at five standard locations on Priest Lake, Idaho from 2001 to 2018.

Year	Cavanaugh Bay	Copper Bay	Huckleberry Bay	Hunt Creek	Indian Creek Bay	Total
2001	523	588	200	232	222	1,765
2002	921	549	49	306	0	1,825
2003	933	1,237	38	624	0	2,832
2004	1,673	1,584	359	2,060	441	6,117
2005	916	906	120	2,961	58	4,961
2006	972	1,288	43	842	0	3,145
2007	463	308	38	1,296	40	2,145
2008	346	223	0	884	27	1,480
2009	550	400	37	1,635	15	2,637
2010	331	37	18	1,410	49	1,845
2011	1,340	750	90	16,103	1,050	19,333
2012	3,135	7,995	665	14,570	830	27,195
2013	2,295	1,070	340	26,770	1,270	31,745
2014	838	1,960	525	7,530	2,750	13,603
2015	1,155	1,885	7	2,550	520	6,117
2016	710	524	34	2,987	670	4,925
2017	660	415	80	1,340	184	2,679
2018	545	670	0	2,995	185	4,395

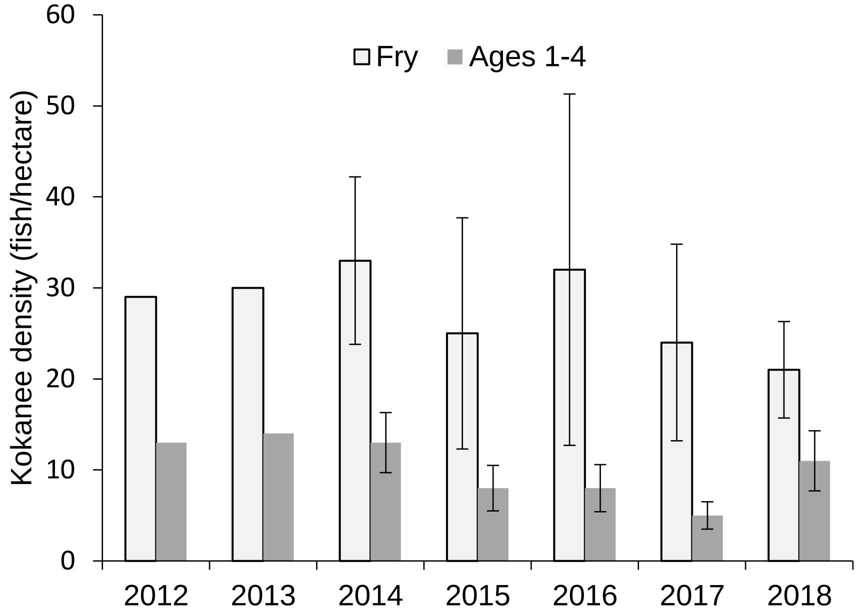

Figure 15. Kokanee density estimates from Priest Lake, Idaho acoustic surveys between 2012 and 2018.

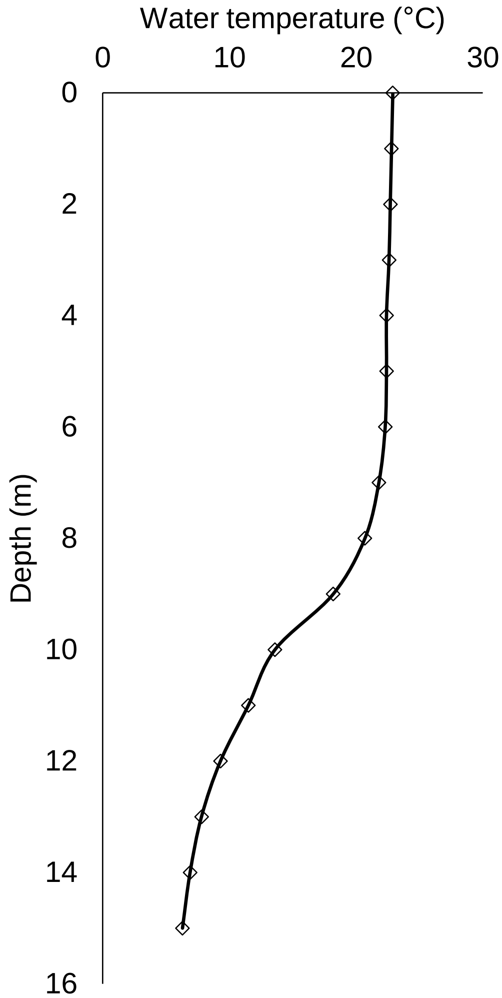

Figure 16. Temperature profile measured in association with our August 2018 acoustic survey of Priest Lake, Idaho.

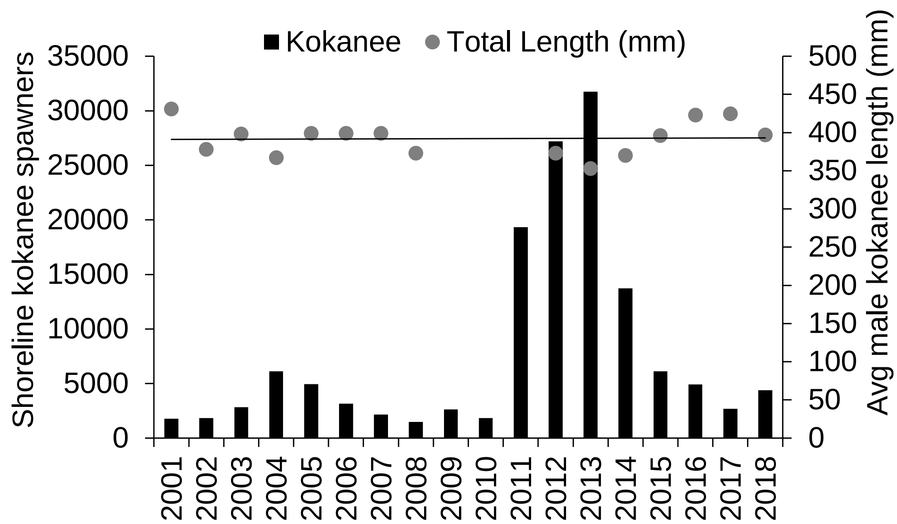

Figure 17. Adult kokanee spawner counts at five standard locations on Priest Lake, Idaho from 2001 through 2018 and corresponding average length of male kokanee spawners.

PRIEST BASIN BULL TROUT EDNA SURVEY

ABSTRACT

Fish communities in the Priest basin have changed over time with observed reductions in native fishes. An inventory of Priest basin tributary fish communities in 2016 suggested Bull Trout *Salvelinus confluentus* were not present in several tributaries where their presence had previously been documented. The probability of detection in fish community surveys is likely low in streams with low abundance and a finer-scale tool may be beneficial to clearly define Bull Trout occurrence. As such, we sampled eDNA in Priest basin tributaries to describe occurrence of Bull Trout and confirm results from recent fish community surveys. We surveyed for Bull Trout eDNA in 14 tributaries of Priest Lake and Upper Priest Lake from August 27, 2018 through October 23, 2018 in collaboration with the Range-Wide Bull Trout eDNA Project. Samples were collected by filtering water through a microfiber filter. All samples were analyzed by the National Genomics Center for Wildlife and Fish Conservation. Bull Trout eDNA was detected at 17 sites among 8 tributaries. Bull Trout eDNA detections in multiple tributaries suggested Bull Trout were more widely distributed than described in a recent electrofishing survey. However, isolated detections of Bull Trout eDNA in most positively sampled tributaries further suggested Bull Trout were not abundant.

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INTRODUCTION

Fish communities in the Priest basin have changed over time. Historically, native fishes (i.e., Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi,* Bull Trout *Oncorhynchus confluentus*) provided robust fisheries in Upper Priest Lake, Priest Lake, and major tributaries to these lakes (Bjornn 1957). Introductions of non-native fishes altered the fish community and the resulting recreational fisheries. Kokanee *Oncorhynchus nerka* and later Lake Trout *Salvelinus namaycush* replaced native fishes as the primary angler targets between the late 1950s and the present (Watkins et al. 2018). Introductions of mysid shrimp *Mysis diluviana* in Priest Lake in the 1960's benefitted Lake Trout abundance and subsequently accelerated declines of other previously abundant fish species, including kokanee, Bull Trout, and Westslope Cutthroat Trout (IDFG 2013, Watkins et al. 2018). Bull Trout have largely been absent in Priest Lake since the 1990's.

Tributaries to Priest Lake and Upper Priest Lake are important spawning and juvenile rearing areas for native and non-native fishes of migratory and resident life histories. Periodic surveys of tributary fish communities have suggested distribution and abundance of native and non-native fishes have changed over time (Bjornn 1957, Irving 1987, DuPont et al. 2008, Ryan et al. 2020b). These investigations and others (Rieman et al. 1979, Mauser et al. 1988, Horner et al. 1988) suggest that angler harvest, habitat degradation, and competition/predation by non-native fishes likely contributed to reductions in abundances of native fishes. Most recently, investigation of tributary fish communities in the Priest basin suggested Westslope Cutthroat Trout were still abundant and commonly represented across surveyed tributaries (Ryan et al. 2020b). However, Bull trout were less abundant than historic levels.

An inventory of Priest basin tributary fish communities in 2016 suggested Bull Trout were not present in several tributaries where their presence had previously been documented (Ryan et al. 2020b). In addition, densities in that survey were low where Bull Trout were detected. Ryan et al. (2020b) suggested a finer-scale tool may be beneficial for describing Bull Trout distribution where the probability of detection is low due to small population size. Sampling of environmental DNA (eDNA) has been demonstrated to be a useful approach for describing occurrence of Bull Trout with fine-scale resolution (McKelvey et al. 2016). In 2018, we sampled eDNA in Priest basin tributaries to describe occurrence of Bull Trout and confirm results from recent fish community surveys.

METHODS

We surveyed for Bull Trout eDNA in 14 tributaries of Priest Lake and Upper Priest Lake from August 27, 2018 through October 23, 2018 (Table 11; Figure 18). Our survey was a collaborative effort with the Range-Wide Bull Trout eDNA Project (Young et al. 2017). Sample sites were pre-determined as a component of the parent project. Generally, sample sites were located in tributary segments where temperature was predicted to be <11°C and having a probability of Bull Trout occurrence > 0.10 (Young et al. 2017). A total of 95 sites were sampled among all streams. Sites per stream varied by stream length incorporated in the survey. Streams surveyed did not represent all locations in the Priest basin where conditions were predicted to be suitable for Bull Trout. In addition, this survey did not target streams where Bull Trout were known to be relatively abundant, such as the Upper Priest River.

Bull Trout eDNA samples were collected from filtered water as described by Carim et al. (2016). A portable electric peristaltic pump was used to pump five liters of water through a microfiber filter to collect present eDNA. After sampling, filters were stored in silica desiccant gel beads to maintain a dry storage environment. Multiple precautions were taken to avoid contamination of the samples, including the use of latex gloves, sterilized sampling equipment, and a detailed sterile sampling protocol. All samples were analyzed by the National Genomics Center for Wildlife and Fish Conservation using methods described by McKelvey et al. (2016). We reported the presence or absence of Bull Trout eDNA for each site.

RESULTS

Bull Trout eDNA was detected at 17 of the 95 sites (18%) sampled, and among 8 of the 14 tributaries. Positive detections were found in Granite, Hunt, Indian, North Fork Indian, Lion, Two Mouth, Trapper, and Floss creeks (Table 11; Figure 18). However, most positive detections represented isolated occurrences near the mouths of most streams. In Granite Creek, two isolated detections occurred, but a single sample site where no detection occurred separated these detections. In Hunt, Lion, Two Mouth, and Trapper creeks, isolated detections were found near the confluence of these streams and the associated lake to which they flow. A single detection was also found in Floss Creek, a tributary of Trapper Creek, isolated high in the drainage. In Indian and North Fork Indian creeks, positive detections were relatively widespread and continuous.

DISCUSSION

Our eDNA samples indicated Bull Trout were only present in 8 of the 14 tributaries sampled. However, eDNA detections in multiple tributaries suggest that Bull Trout were more widely distributed than previously described in a recent electrofishing survey. Ryan et al. (2020b) did not detect Bull Trout in most tributaries that we sampled using eDNA, including Granite, Hunt, Indian, Lion, Two Mouth, and Trapper creeks. Electrofishing samples in that survey did detect Bull Trout in North Fork Indian Creek. An electrofishing survey was not completed on Floss Creek. The disparity between surveys highlights the limitations of electrofishing surveys for describing the occurrence of a species when population density is low.

While we detected Bull Trout in 8 of the 14 tributaries sampled, most were isolated detections near the tributary mouths, suggesting Bull Trout were not abundant in a majority of streams sampled. Our observations contrasted modeled habitat availability that indicated suitable habitat was widespread in most of the sampled drainages (Young et. al 2017). We saw positive eDNA detections in only one or two locations among a wider distribution of samples in all positive tributaries, except the Indian and North Fork Indian creeks. The detection probability of eDNA in flowing water is generally positively related to abundance and negatively related to distance from the sample subject (Wilcox et al. 2016). This suggests isolated positive samples within a distribution of samples likely capture the presence of few fish in a confined area. The location of positive eDNA detections in Hunt, Lion, Two Mouth, and Trapper creeks were low in the drainage near the tributaries confluence with the lake. We hypothesize that these detections may represent transient individuals rather than populations consisting of multiple fish. If so, it is possible that reproduction is not occurring in these tributaries.

The distribution of positive Bull Trout eDNA detections in the Indian Creek drainage may overemphasize the true level of Bull Trout distribution in that tributary. Ryan et al. (2020b), observed Bull Trout in a single location on North Fork Indian Creek in their electrofishing survey. Bull Trout density in that sampling location was moderate. However, distribution high in the drainage and low in the drainage (i.e., Indian Creek) was not evident in their work. In contrast, positive eDNA detections occurred at multiple sequential sites on North Fork Indian Creek and downstream throughout Indian Creek. We assume eDNA drift in the drainage may have extended the distribution of positive eDNA detections beyond locations where Bull Trout were physically located. Although interpretation of these two surveys may not overlap completely, we suggest both provide evidence that Bull Trout are more abundant and widely distributed in the Indian Creek drainage than other sampled Priest Lake tributaries. Ryan et al. (2020b) also suggested this population was unique relative to other tributaries to Priest Lake. They found Bull Trout size structure in North Fork Indian Creek included larger individuals than typically observed in juveniles of adfluvial populations and suggested a resident population may occur in that location. Combined, these observations imply this population may differ from others in the drainage and may pose unique management implications in addressing long-term conservation of Bull Trout.

This investigation provided further understanding of the occurrence of Bull Trout in the Priest basin. Although this work provided information for a wide selection of locations, information gaps remain and may warrant further investigation. For example, we did not investigate the occurrence of Bull Trout in tributaries to Granite Creek. Our investigation of Granite Creek provided some evidence that Bull Trout were present, but with a limited distribution. We noted positive eDNA detections in Granite Creek occurred in the vicinity of several tributaries. We found no information describing fish communities in adjacent tributaries to Granite Creek, but speculate Bull Trout presence in these tributaries could have influenced our findings in Granite Creek. Brook Trout are widespread in Granite Creek and may further influence the distribution of any remnant Bull Trout population in the drainage (Dupont et al. 2008). We recommend some reconnaissance of Granite Creek tributaries be completed to improve our understanding of species occurrences in the drainage.

RECOMMENDATIONS

1. Complete reconnaissance of Granite Creek tributaries to improve our understanding of species occurrences in the drainage.

Table 11. Bull Trout eDNA detections by stream and site from surveys of Priest Lake and Upper Priest Lake tributaries in 2018.

Stream	Date collected	Site ID	Latitude	Longitude	BLT detected
Granite Creek	9/6/2018	544-1	48.677490	-116.974370	0
Granite Creek	9/6/2018	544-2	48.684060	-116.980690	0
Granite Creek	9/6/2018	547-1	48.688660	-116.984530	1
Granite Creek	9/6/2018	547-2	48.688620	-116.996190	0
Granite Creek	9/6/2018	555-1	48.691690	-117.002560	1
Granite Creek	9/6/2018	558-1	48.697910	-117.013310	0
Granite Creek	9/6/2018	564-1	48.698840	-117.025630	0
Hunt Creek	10/22/2018	384-1	48.566890	-116.824550	1
Hunt Creek	10/22/2018	387-1	48.564010	-116.828870	0
Hunt Creek	10/23/2018	389-1	48.584790	-116.754640	0
Hunt Creek	10/23/2018	389-2	48.585000	-116.741270	0
Hunt Creek	10/22/2018	391-1	48.566300	-116.808570	0
Hunt Creek	10/22/2018	391-2	48.572556	-116.798082	0
Hunt Creek	10/22/2018	391-3	48.578100	-116.790210	0
Hunt Creek	10/22/2018	393-1	48.581990	-116.774220	0
Hunt Creek	10/23/2018	393-2	48.585360	-116.762160	0
South Fork Hunt Creek	10/23/2018	368-1	48.565090	-116.777370	0
South Fork Hunt Creek	10/23/2018	368-2	48.563210	-116.764140	0
South Fork Hunt Creek	10/23/2018	368-3	48.561300	-116.751590	0
Indian Creek	9/4/2018	439-1	48.611400	-116.836200	1
Indian Creek	9/4/2018	439-2	48.619640	-116.831130	1
Indian Creek	8/29/2018	446-1	48.632320	-116.808430	1
Indian Creek	8/29/2018	446-2	48.633510	-116.796050	1
Indian Creek	9/4/2018	448-1	48.624680	-116.828640	1
Indian Creek	9/4/2018	448-2	48.628210	-116.819040	1
North Fork Indian Creek	8/28/2018	421-1	48.658730	-116.718220	0
North Fork Indian Creek	8/28/2018	421-2	48.658840	-116.705360	0
North Fork Indian Creek	8/28/2018	421-3	48.655070	-116.693470	0
North Fork Indian Creek	8/28/2018	421-4	48.649490	-116.687280	0
North Fork Indian Creek	8/29/2018	451-1	48.634360	-116.789590	1
North Fork Indian Creek	8/29/2018	451-2	48.641040	-116.780560	1
North Fork Indian Creek	8/29/2018	452-1	48.644710	-116.773160	1
North Fork Indian Creek	8/28/2018	464-1	48.646390	-116.760730	1
North Fork Indian Creek	8/28/2018	464-2	48.651230	-116.748790	0
North Fork Indian Creek	8/28/2018	465-1	48.656120	-116.731870	0
South Fork Indian Creek	9/6/2018	428-1	48.625510	-116.770400	0
South Fork Indian Creek	9/6/2018	432-2	48.628660	-116.778120	0
Lamb Creek	8/27/2018	427-1	48.535110	-117.003620	0
Lamb Creek	8/27/2018	427-2	48.535880	-117.015360	0
Lamb Creek	8/27/2018	427-3	48.533110	-117.027430	0
Lamb Creek	8/27/2018	427-4	48.541770	-117.031590	0
Lamb Creek	8/27/2018	427-5	48.547210	-117.041680	0
Lamb Creek	8/27/2018	427-6	48.551550	-117.053110	0
Lamb Creek	8/29/2018	427-7	48.556830	-117.062910	0
Lamb Creek	8/29/2018	427-8	48.558070	-117.075260	0

Table 11 (continued)

Stream	Date collected	Site ID	Latitude	Longitude	BLT detected
Lion Creek	9/24/2018	536-1	48.760030	-116.666610	0
Lion Creek	9/24/2018	536-2	48.756930	-116.662820	0
Lion Creek	9/24/2018	571-1	48.763650	-116.703490	0
Lion Creek	9/24/2018	571-2	48.763840	-116.691890	0
Lion Creek	9/24/2018	571-3	48.764140	-116.679120	0
Lion Creek	9/13/2018	575-1	48.743630	-116.815140	0
Lion Creek	9/13/2018	575-2	48.741810	-116.803380	0
Lion Creek	9/24/2018	576-1	48.743450	-116.796200	0
Lion Creek	9/24/2018	577-1	48.745540	-116.785720	0
Lion Creek	9/24/2018	579-1	48.737090	-116.829770	0
Lion Creek	9/13/2018	579-2	48.740610	-116.823170	1
Lion Creek	9/13/2018	580-1	48.748090	-116.775180	0
Lion Creek	9/13/2018	580-2	48.750220	-116.763020	0
Lion Creek	9/13/2018	581-1	48.753610	-116.753350	0
Lion Creek	9/13/2018	582-1	48.753970	-116.752130	0
Lion Creek	9/24/2018	586-1	48.763060	-116.715710	0
Lion Creek	9/13/2018	587-1	48.756950	-116.732500	0
Lion Creek	9/24/2018	591-1	48.760880	-116.724080	0
South Fork Lion Creek	9/24/2018	527-3	48.734440	-116.777740	0
South Fork Lion Creek	9/24/2018	527-4	48.734160	-116.766950	0
Trapper Creek	10/9/2018	676-1	48.797240	-116.895200	1
Trapper Creek	10/10/2018	676-2	48.803740	-116.894680	0
Trapper Creek	10/8/2018	676-3	48.811090	-116.891090	0
Trapper Creek	10/9/2018	686-1	48.817720	-116.893350	0
Trapper Creek	10/9/2018	697-1	48.826250	-116.894410	0
Trapper Creek	10/4/2018	707-1	48.837760	-116.881130	0
Trapper Creek	10/4/2018	723-1	48.848480	-116.880220	0
Trapper Creek	10/4/2018	723-2	48.857480	-116.880400	0
East Fork Trapper Creek	10/9/2018	677-1	48.817510	-116.892670	0
Floss Creek	10/19/2018	655-1	48.810130	-116.876720	0
Floss Creek	10/20/2018	655-2	48.804540	-116.867850	0
Floss Creek	10/19/2018	671-1	48.817860	-116.884450	0
Floss Creek	10/19/2018	675-3	48.820180	-116.856790	1
Two Mouth Creek	9/17/2018	476-2	48.692410	-116.679560	0
Two Mouth Creek	9/17/2018	502-1	48.701650	-116.684940	0
Two Mouth Creek	9/17/2018	505-1	48.696460	-116.724120	0
Two Mouth Creek	9/17/2018	505-2	48.699170	-116.711420	0
Two Mouth Creek	9/17/2018	505-3	48.701190	-116.698590	0
Two Mouth Creek	9/17/2018	506-1	48.698060	-116.747290	0
Two Mouth Creek	9/17/2018	506-2	48.696840	-116.733630	0
Two Mouth Creek	9/17/2018	509-1	48.698960	-116.754550	0
Two Mouth Creek	9/12/2018	511-1	48.697630	-116.761960	0
Two Mouth Creek	9/12/2018	512-1	48.697500	-116.766230	0
Two Mouth Creek	9/12/2018	513-1	48.696310	-116.785000	0
Two Mouth Creek	9/12/2018	514-1	48.692920	-116.800280	0
Two Mouth Creek	9/12/2018	515-1	48.695940	-116.822310	0
Two Mouth Creek	9/12/2018	515-2	48.692290	-116.811090	0

Table 11 (continued)

Stream	Date collected	Site ID	Latitude	Longitude	BLT detected
Two Mouth Creek	9/12/2018	516-1	48.693740	-116.797450	0
Two Mouth Creek	9/25/2018	521-1	48.690040	-116.831240	1
Two Mouth Creek	9/12/2018	521-2	48.695220	-116.828440	0

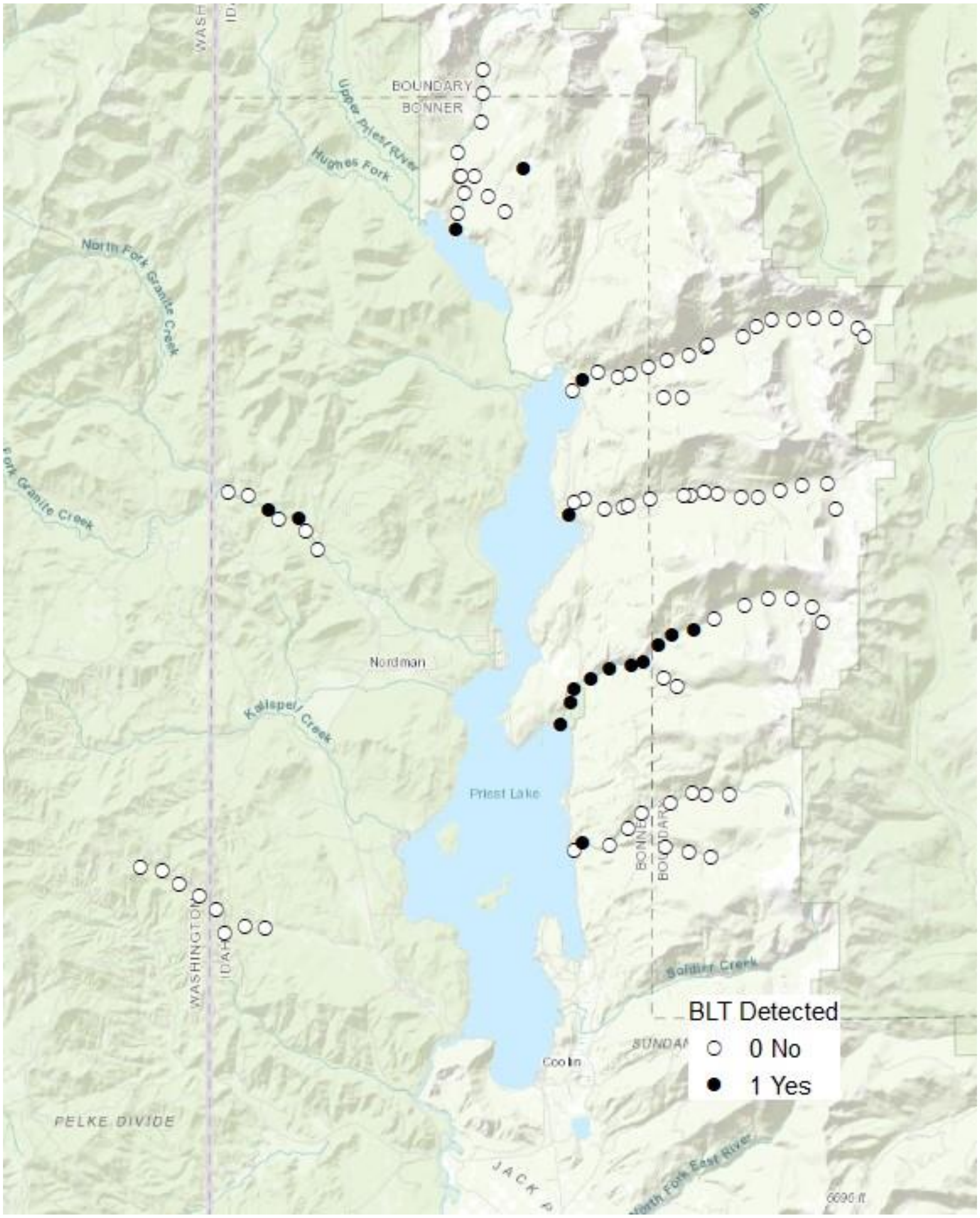

Figure 18. Environmental DNA (eDNA) survey locations on tributaries to Priest Lake and Upper Priest Lake, Idaho sampled in 2018. White circles represent survey locations where Bull Trout eDNA was not detected. Black circles represent survey locations where Bull Trout eDNA was detected.

PEND OREILLE RIVER LITTORAL FISH COMMUNITY ASSESSMENT

ABSTRACT

In 2016, we surveyed the Pend Oreille River littoral fish community with an interest in describing current species composition, relative abundance, and trends in abundance and size structure. We also evaluated the use of minimum length limits for increasing the number of large (≥ 400 mm) Smallmouth Bass *Micropterus dolomieu* in the fishery. We completed a survey of the Pend Oreille River littoral fish community from May 30 to June 2. Boat-mounted electrofishers were used to sample fish. Angler exploitation of Smallmouth Bass was estimated by tagging a subsample of fish collected in our survey. We applied a Beverton-Holt yield-per-recruit model in FAMS to evaluate the effect of harvest on abundance of large (≥ 400 mm) Smallmouth Bass. We collected 18 fish species among all sample sites in our survey. Yellow Perch *Perca flavescens* were the most abundant species caught (72.8 ± 14.8 fish/h; CPUE ± 80% C.I.) and represented 30% of the catch and 9% of the biomass. Smallmouth Bass (36.6 ± 8 fish/h), Peamouth *Mylocheilus caurinus* (34.6 ± 6.9 fish/h), Black Crappie *Pomoxis negromaculatus* (29.9 ± 6.9 fish/h), and Pumpkinseed *Lepomis gibbosus* (29.7 ± 9.9 fish/h) were also well represented. Largemouth Bass *Micropterus salmoides* were poorly represented at 2% of the catch and a catch rate of 4.2 fish/h (± 1.8). We estimated that annual mortality of Smallmouth Bass ages 2-9 was 51%. Angler exploitation of Smallmouth Bass was low (8%). We detected differences in CPUE over time for most species sampled, but a common trend across species was not observed. Our model predicted that application of a 400-mm minimum length harvest restriction would only minimally increase the proportion of the Pend Oreille River Smallmouth Bass population achieving 400 mm, except under very low natural mortality levels.

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INTRODUCTION

The Pend Oreille River originates at the outflow of Lake Pend Oreille in northern Idaho. It flows west through the Idaho Panhandle into Washington, north through northeastern Washington into British Columbia, and west to its confluence with the Columbia River. Approximately 26 miles of the river occur within Idaho. Albeni Falls Dam, located on the Pend Oreille River near the border of Idaho and Washington, regulates water levels in the river and Lake Pend Oreille. Water elevation in the river and lake seasonally fluctuates up to 11.5 ft between summer full pool and winter drawdown.

The fish community of the Pend Oreille River is diverse, including both a variety of native and non-native fishes. Species present include Black Crappie *Pomoxis negromaculatus*, Brook Trout *Salvelinus fontinalis*, Brown Bullhead *Ameiurus nebulosus*, Brown Trout *Salmo trutta*, Kokanee *Oncorhynchus nerka*, Lake Whitefish *Coregonus clupeaformis*, Largemouth Bass *Micropterus salmoides*, Largescale Sucker *Catostomus macrocheilus*, Longnose Sucker *Catostomus catostomus*, Northern Pikeminnow *Ptychocheilus oregonensis* , Peamouth *Mylocheilus caurinus*, Pumpkinseed *Lepomis gibbosus*, Rainbow Trout *Oncorhynchus mykiss*, Smallmouth Bass *Micropterus dolomieu*, Tench *Tinca tinca*, Walleye *Stizostedion vitreum*, Westslope Cutthroat Trout *Oncorhynchus clarkii*, and Yellow Perch *Perca flavescens* (Bennett and Dupont 1993, Maiolie et al. 2011). Redside Shiner *Richardsonius balteatus*, once common in the system are now rare (Maiolie et al. 2011). Species composition and abundance in the Pend Oreille River are thought to be heavily influenced by water level management (Bennett and Dupont 1993, Schoby et al. 2007, Maiolie et al. 2011). Unintended introduction of new fish species (i.e., Smallmouth Bass, Walleye) has also influenced composition and abundance within the fish community (Schoby et al. 2007, Maiolie et al. 2011).

Currently, Pend Oreille River fisheries are managed under both general regional fishing regulations and special exceptions to the general regulations. Exceptions apply to Largemouth Bass and Westslope Cutthroat Trout. Largemouth Bass harvest is regulated under a two-fish bag limit; none may be less than 406 mm. The current regulation on Largemouth Bass was initiated in 2008 on the Pend Oreille River and extended more widely to adjacent slough habitats in 2011. No harvest opportunity is available for Westslope Cutthroat Trout. Required catch-and-release of Westslope Cutthroat Trout was initiated on the Pend Oreille River and Lake Pend Oreille beginning in 2001. A general bag limit of six fish of any size currently applies to Smallmouth Bass in the system, although the bag limit is in combination with Largemouth Bass. A desire for more conservative regulations has been expressed by some anglers with an interest in increasing the abundance of large Smallmouth Bass.

In 2016, we surveyed the Pend Oreille River littoral fish community with an interest in describing current species composition, relative abundance, and general trends in abundance and size structure. In addition, we used species-specific information to inform angling regulations for managing recreational fisheries. Specifically, we evaluated whether minimum length limits could increase the abundance of large Smallmouth Bass in the fishery.

OBJECTIVES

1. Describe current conditions and trends in the Pend Oreille River littoral fish community.

2. Evaluate the effect of a minimum length limit harvest restriction for increasing abundance of large (≥ 400 mm) Smallmouth Bass in the Pend Oreille River.

METHODS

We completed a Pend Oreille River littoral fish community assessment from May 30 to June 2, 2016. Sampling was conducted using a simple random survey design. Fifty random sample sites were chosen *a priori*. Sample sites were distributed within the Idaho portion of the river between the boundaries of the U.S. Hwy 95 "Long Bridge" and Albeni Falls Dam (Table 12). The river shoreline was divided into unique numbered segments from which sample sites were selected. Segments were created by overlaying a 1,000-m grid on a map of the Pend Oreille River using Terrain Navigator Pro (My Topo; Billings, Montana).

Boat mounted electrofishers were used to sample fish. We used one or two Smith-Root (Smith-Root; Vancouver, WA) electrofishing boats per night to complete sampling efforts. Boats included a 4.8 m Smith-Root 5.0 GPP and a 6.1 m Smith-Root 7.5 GPP electrofishing boats. We used DC current of 60 pulses per second in a high voltage setting (i.e. > 500 volts). We adjusted duty cycle periodically during each sampling event to maximize fish attraction while limiting mortality. Two people per boat attempted to net all fish during each sample unit. Sample units were ten minutes in duration. Each unit began at a pre-determined sample site and generally proceeded in an upstream direction.

All fish collected were identified, measured to total length (mm), weighed (g), and released unless sacrificed for removal of ageing structures. Relative abundance was reported as average catch per unit effort (CPUE). CPUE was standardized to catch per hour for reporting. We described the general structure of the fish community as the relative percentage of each species and relative percentage of biomass of each species in the sample. Size structure of sampled species was described using length-frequency histograms and proportional stock density (PSD) indices (Anderson and Neumann 1996) for primary sportfish species. Relative stock density of preferred size Smallmouth Bass (RSD-P) was also calculated. We used Fisheries Analysis and Modeling Simulator (FAMS, Slipke and Maceina 2014) software to calculate stock density indices. Relative weight (W_r, Wege and Anderson 1978) was used to describe the condition of primary sportfish species.

Otoliths were removed from a subsample of Smallmouth Bass for age estimation. We also removed dorsal spines for age estimation from a subsample of Largemouth bass. We targeted three to five ageing structures per centimeter group per species. Otoliths were broken centrally across the transverse plane, browned on the broken surface with a lighter, sanded to improve viewing, and viewed under 10x to 30x magnification on a dissecting microscope. Dorsal spines were mounted in epoxy, sectioned near the proximal end on a Buehler Isomet saw (Illinois Tool Works Inc., Lake Bluff, Illinois), sanded for viewing clarity, and viewed on a compound microscope under 40x to 100x magnification. Length-at-age at time of capture was reported as an index of growth where applicable.

We used Smallmouth Bass length and age data to estimate rates of growth and mortality. Growth rates were described as von Bertalanffy growth coefficients, estimated in FAMS from mean values of total length-at-age observed in our sample. Catch-at-age of sampled Smallmouth Bass was used to describe general patterns of recruitment and to estimate mortality rates. An age-length key was used to predict ages of Smallmouth Bass based on length from a subsample of age estimates. Age frequencies were applied to a weighted catch curve generated in FAMS to estimate instantaneous total mortality (Z), from which annual mortality (A) and annual survival (S) were derived. Confidence intervals (80%) around Z were estimated from the mean square error of the regression model used to estimate Z as described in Miranda and Bettoli (2007). Few

Largemouth Bass were collected, thus preventing accurate estimates of annual mortality rates for that species.

Angler exploitation of Smallmouth Bass was estimated using a subsample of fish collected in our survey. We tagged and released Smallmouth Bass 254 mm and greater with individually numbered T-bar style tags (Floy, Inc.). Tags were inserted at an angle into the dorsal musculature just below the dorsal fin of each fish. Sample size was enhanced by tagging additional fish on several dates from June 14 through June 23. Targeted tagging efforts were distributed throughout the Pend Oreille River, but were focused in areas we anticipated catching Smallmouth Bass. Each tag was printed with the Idaho Department of Fish and Game "Tag You're It" phone number for reporting. No reward was offered for tag returns. Angler tag returns were collected by phone, online (IDFG website), and in person at the IDFG Panhandle Regional Office through January 2017. Exploitation rates were estimated using tag returns as described by Meyer et al. (2012). We corrected tag returns for tag loss (10.5%), tagged fish mortality (2.0%), and reporting rate (55%; Meyer et al. 2012, Meyer and Schill 2014).

The information gathered from our survey was used to evaluate trends in the Pend Oreille River fish community. We evaluated trends by comparing relative abundance to prior surveys of the Pend Oreille River completed in 2005 and 2010 (Schoby et al. 2007, Maiolie et al. 2011). To allow for meaningful comparisons with 2016 survey results, we transformed CPUE estimates and 80% confidence bounds from 2005 and 2010 spring electrofishing data to fish caught per hour. Prior surveys sampled primarily in ten minute effort units, but reported CPUE as cumulative fish/minute and were not able to estimate confidence bounds around CPUE estimates. Differences in CPUE between years were tested using one way analysis of variance (α = 0.20). Differences between years were described using a Tukey's *post hoc* evaluation. Statistical tests were run in SYSTAT (Systat Software Inc.). We also compared population characteristics of primary sport fishes between surveys where metrics were available. Compared metrics included length-at-age at time of capture as a measure of growth, PSD as a measure of size structure, and annual mortality. Estimates of Smallmouth Bass RSD-P were also compared between survey years.

Water level manipulations of the Pend Oreille River have previously been suggested to influence both survival and recruitment of warmwater fishes (Bennett and Dupont 1993, Schoby et al. 2007). Schoby et al. (2007) suggested years in which Lake Pend Oreille and the Pend Oreille River experienced spring refill to an elevation of 628 m by early- to mid-May produced stronger cohorts of Largemouth Bass and Smallmouth Bass. To investigate how spring water levels may have influenced year class strength of warmwater fishes, we summarized the date at which Lake Pend Oreille water levels reached an elevation of 628 m. We used Lake Pend Oreille water levels as a surrogate for Pend Oreille River levels because no gauge was available above Albeni Falls Dam on the Idaho portion of the river. Historic Lake Pend Oreille water level data were taken from United States Geological Service archives (https://waterdata.usgs.gov). We also summarized average January water levels in Lake Pend Oreille to investigate how winter water levels may have influenced warmwater fishes. We did not sample enough Largemouth Bass to adequately investigate water year effects on abundance.

Regulation Modeling

We applied a Beverton-Holt yield-per-recruit model to evaluate the effect of harvest on abundance of large (≥ 400 mm) Smallmouth Bass in the Pend Oreille River. FAMS was used to develop and run models. Primary model inputs included growth and mortality rates estimated from data collected in our survey (Table 13).

Fish growth was incorporated into our model in the form of linear coefficients of a length-weight relationship and von Bertalanffy growth coefficients, both estimated from analysis of our 2016 survey data. Although we used 2016 survey data to estimate growth coefficients, we incorporated length information from a previous survey to improve the accuracy of growth coefficients. Electrofishing capture of Smallmouth Bass is known to be biased negatively against length (Beamesderfer and Rieman 1988). Maximum length of Smallmouth Bass in our 2016 survey was shorter than described in a 2014 fall Walleye index gillnetting survey of the Pend Oreille River and Lake Pend Oreille (Watkins et al. 2018). As such, we used maximum length observed in the 2014 survey to define the theoretical maximum length of Smallmouth Bass (L_∞) in calculations of von Bertalanffy growth coefficients. We held L_∞ constant at 510 mm.

Our model incorporated both conditional fishing (*cf*) and conditional natural (*cm*) mortality rates. Conditional mortality rates (*cf* and *cm*) were calculated as described in Miranda and Bettoli (2007) and Slipke and Maceina (2000). We used a range of mortality rates rather than site-specific values to address uncertainty in rate estimates from our survey data. Conditional natural mortalities varied from 20% to 50% to simulate high and low estimates of *Z*. We estimated *Z* using a catch curve as described previously. Our estimate of *Z* was assumed to be positively biased due to selectivity of electrofishing gear on Smallmouth Bass. As such, we used our central estimate as a high-level mortality rate. The lower bound of our estimate of *Z* was applied as our low-level mortality rate. We confirmed mortality rates used in the model were reasonable by estimating instantaneous natural mortality (*M*) and *cm* using five computational mortality estimators available in FAMS. These estimators relied on population characteristics, such as maximum age and growth to predict natural mortality. Estimates of *M* were only used to provide a general reference of the range of natural mortality rates possible. Conditional fishing mortalities varied from 10% to 30% and corresponded to approximate exploitation levels of 7% to 27%. Conditional fishing mortality values were based on estimates of exploitation of Smallmouth Bass from the Pend Oreille River in 2016 and 2006 (see this report, Dupont et al. 2009). Both referenced exploitation estimates were low (≤ 15%). However, we incorporated higher levels of *cm* into the model to account for uncertainty around fishing-related mortality and to observe the potential effect of increased fishing mortality.

We applied length-specific fishing mortality within the model to simulate two harvest scenarios, including the absence of a length limit and a 400 mm minimum length limit. We simulated the no length limit scenario by applying *cm* rates to fish 254 mm and greater. We assumed that without a length limit anglers would not be willing to harvest Smallmouth Bass less than 254 mm. A 400-mm minimum length limit was simulated by applying *cm* only to fish 400 mm and greater. We applied fishing mortalities uniformly across designated size ranges. True fishing mortality may not be uniform across length, but adequate size-specific harvest information was not available.

Our model was used to describe the impact of a 400-mm length limit relative to no length limit. Model outputs included the proportion of a cohort reaching 400 mm. Model outputs were compared by describing the difference in proportions of a cohort reaching 400 mm between the two modeled regulations.

RESULTS

We collected 18 fish species among all sample sites in our 2016 survey of littoral habitats of the Pend Oreille River (Table 14). Yellow Perch were the most abundance species caught (CPUE ± 80% C.I.; 72.8 ± 14.8 fish/h) representing 30% of the catch and 9% of the biomass. Smallmouth Bass (36.6 ± 8 fish/h), Peamouth (34.6 ± 6.9 fish/h), Black Crappie (29.9 ± 6.9 fish/h), and Pumpkinseed (29.7 ± 9.9 fish/h) were also well represented. Largemouth Bass were poorly represented at 2% of the catch and a catch rate of 4.2 fish/h (± 1.8). Other sportfish including salmonids, Lake Whitefish, and Walleye were also poorly represented, although we expected the nature of survey methods (i.e., spring electrofishing in littoral habitats) to generally be less effective at sampling these species.

Sampled Yellow Perch varied in total length from 70 to 265 mm (Table 14; Figure 19). We estimated PSD of the collected sample at 3.5, which indicated few fish were larger than quality length (200 mm; Table 15). Mean relative weight was estimated at 78.

Largemouth Bass in our sample varied in total length from 65 to 520 mm (Table 14; Figure 19). PSD of sampled fish was estimated at 45 (Table 15). Largemouth Bass were generally robust with a mean relative weight of 100. Too few individuals (*n* = 35) were collected to make meaningful conclusions regarding growth or mortality.

Smallmouth Bass PSD was 20.8 demonstrating the majority of the catch was smaller than quality length (280 mm; Table 15, Figure 19). Total length of collected fish varied from 52 to 480 mm (Table 14). Smallmouth Bass RSD-P was 7. The von Bertalanffy growth coefficients were estimated as K = 0.182 and t_0 = -0.862 with L_∞ held constant at 510 mm. We estimated Smallmouth Bass grew to 305 mm in approximately 4.1 years (Figure 20). Mean W_r was 86.

We estimated Z of Smallmouth Bass from two to nine years of age at -0.72 (± 0.3, 80% C.I.; Figure 21). Corresponding A was 51%. The mean of instantaneous natural mortality estimators was M = 0.43 with a corresponding *cm* of 0.35. We observed fish in our sample from 1-9 years of age (Figure 21). Year class strength appeared to be variable. Specifically, we observed large shifts in catch between age-2 and age-3, as well as 4 and 5 (Figure 21).

Angler exploitation of Smallmouth Bass was estimated to be low. Nineteen of 151 tagged fish were reported caught by anglers. Only six of those fish reported were harvested. We estimated adjusted exploitation at 8%. Harvested Smallmouth Bass reported by anglers represented a range of lengths generally spanning the length distribution of tagged fish (Figure 22).

Differences in species-specific CPUE estimates were detected among survey years for most species sampled ($P \leq 0.20$; Table 16, Figure 23). Although differences were detected, no common pattern of change was observed. The level of variation in CPUE estimates between surveys varied widely. Large shifts were observed in catches of Black Crappie, Northern Pikeminnow, Pumpkinseed, and Yellow Perch. We observed a significant decline in catch of Largemouth Bass from prior surveys, but did not observe significant differences in catch rates of Smallmouth Bass between any years.

We observed variability in other characteristics of selected warmwater fishes that also suggested populations were different than observed in prior surveys of the Pend Oreille River. Specifically, PSD values declined for Black Crappie, Largemouth Bass, Smallmouth Bass, and Yellow Perch from conditions observed in 2010 and suggested an increase in the proportion of smaller individuals (Table 15). PSD values in 2005 for Black Crappie, Largemouth Bass, and Yellow perch were also lower than 2010 surveys.

Other population characteristics of primary warmwater sportfish in the Pend Oreille River remained relatively constant across our comparison of surveys (Table 15). Average relative weight estimates of Black Crappie, Largemouth Bass, Smallmouth Bass, and Yellow Perch varied only 8% to 13% between 2005, 2010, and 2016 surveys where information was available. Estimated age at 305 mm varied from 4 to 4.6 years of age for Smallmouth Bass. Estimates of *A* varied more widely, but in general Smallmouth Bass exhibited what would typically be considered high annual mortality (51%-69%), and Largemouth Bass had moderate levels of annual mortality (35% to 45%). Smallmouth Bass RSD-P was estimated at 3%, 9%, and 7% in 2005, 2010, and 2016, respectively.

We found both spring fill and winter draw down of Lake Pend Oreille to be variable. The date of spring fill to an elevation of 628 m varied from May 25 to June 6 in the nine years preceding our survey (Figure 24). Average January water elevation in the system varied from 625 to 627 m (Figure 25).

Regulation Modeling

Our model predicted that the application of a 400-mm minimum length limit increased the proportion of Smallmouth Bass achieving 400 mm under most modeled scenarios. When natural mortality was low to moderate (*cm* 0.20 to 0.40) abundance was estimated to be 1% to 15% greater under a 400-mm minimum length limit within the range of evaluated exploitation rates (Figure 26). When natural mortality was high (*cm* = 0.50) little to no benefit was predicted (Figure 26). Fishing mortality was most influential at low natural mortality rates. Increasing exploitation rates influenced population response up to 8% under low natural mortality. Fishing mortality was less influential under moderate to high natural mortality rates, with a maximum of a 2% increase in abundance between low and high exploitation rates.

DISCUSSION

A variety of factors effecting both survival and recruitment likely influenced abundance of Pend Oreille River fishes. For example, a combination of habitat-related conditions have been investigated and linked to year-class strength. Bennett and Dupont (1993) suggested winter draw down of the Pend Oreille River reduced overwinter habitat significantly and likely decreased overwinter survival as a result. Schoby et al. (2006) suggested the timing of water elevation increases in the spring influenced recruitment success of warmwater fishes (i.e., Largemouth Bass, Smallmouth Bass, Black Crappie). Natural variability or cyclic recruitment is a common phenomenon observed in Black Crappie and Yellow Perch populations independent of water-level manipulations (Hooe 1991, Allen and Miranda 2001, Sanderson et al. 1999). Strong year classes of Black Crappie and Yellow Perch were detected in length frequencies from our survey and likely reflected positive recruitment conditions in recent prior years. Abundance of other fish species (e.g., Northern Pikeminnow and Redside Shiner) collected in Pend Oreille River surveys past and present exhibited significant negative trends, likely reflecting larger shifts in species composition. For some species, negative trends in abundance may be related to shifting dynamics as a result of newly introduced species. Maiolie et al. (2011) suggested increasing Smallmouth Bass abundance in the Pend Oreille River negatively affected abundance of Northern Pikeminnow and Redside Shiner.

The cause of declining Largemouth Bass abundance observed between Pend Oreille River surveys was not clear. Gear and survey timing biases are common challenges in fisheries surveys and can affect the interpretation of survey results. The timing and sampling methods associated with our survey were standardized, thus minimizing the potential impact of sampling bias on observed catch rates of Largemouth Bass and other species. Schoby et al. (2007) suggested Largemouth Bass movement from the Pend Oreille River into adjacent isolated slough habitats may be influenced by water levels at the time of the survey. We found water levels during our survey were similar to water levels in previous survey years, suggesting sampling conditions were not a likely factor in differing CPUE rates between surveys. Fishing mortality was also not likely a factor in declining Largemouth Bass abundance. A conservative minimum length limit and bag limit of two fish, none less than 406 mm, was extended throughout the Pend Oreille River and adjacent sloughs in 2011. Exploitation of Largemouth Bass in the 2010-2011 time period was estimated to be low at approximately 5% (Maiolie et al. 2011). We also did not find strong patterns in water elevation either in spring or winter that clearly suggested water-level management impacted warmwater fish abundance differently than observed in prior surveys.

In contrast to our observed decline in Largemouth Bass abundance, Smallmouth Bass abundance in the Pend Oreille River was stable over time. Smallmouth Bass were rare in the Pend Oreille River in the early 1990s (Bennett and Dupont 2003). By 2005, abundance had increased dramatically (Schoby et al. 2007). Prior investigators speculated that Smallmouth Bass abundance would increase rapidly beyond observed abundances (Schoby et. al 2007, Maiolie et. al 2011), but after a decade estimates of abundance have remained consistent. We found no significant changes in relative abundance between our survey and prior surveys in 2005 and 2010 (Schoby et. al 2007, Maiolie et. al 2011). As previously mentioned, winter drawdown of the Pend Oreille River is thought to reduce fish survival and may limit the potential for Smallmouth Bass population growth. In general, Pend Oreille River Smallmouth Bass abundance was low in 2016 relative to the range of abundances observed in other regional and state waters. For example, Watkins et al. (2018) found Smallmouth Bass catch rates were greater (CPUE = 44 fish/hour ± 10, 80% C.I.) in Priest Lake, another recently established population in northern Idaho. Hayden Lake Smallmouth Bass CPUE was approximately 90 fish/hour (IDFG unpublished data). Relative abundance of Smallmouth Bass in Milner Reservoir, a Snake River impoundment in southern Idaho, was considerably greater at 252 fish/hour (Ryan et al. 2008).

Our assessment of the Pend Oreille River littoral fish community suggested significant changes in abundance of multiple fish species occurred over time. Shifts in abundance both positive and negative have implications relative to angling opportunities in the river. An understanding of trends in abundance is important to identify what management actions, if any, are available to influence desirable outcomes. As such, we recommend continued periodic monitoring of the Pend Oreille River littoral fish community occur at five- to ten-year intervals.

Regulation Modeling

Our modeling suggested that minimum length limits applied to the Pend Oreille River Smallmouth Bass fishery would have minimal impact on the abundance of larger fish in the system. We found abundance to be influenced primarily by growth and natural mortality rates, rather than fishing mortality within the range of fishing mortalities estimated for the Pend Oreille River. The modeled minimum length limit represented a conservative harvest regulation. As such, more liberal regulation scenarios (e.g., slot limit) would have less influence on abundance of large fish in the population. Our results were consistent with expectations of a low productivity population. Beamesderfer and North (1995) found that low productivity Smallmouth Bass populations exhibiting high to moderate natural mortality rates and slow growth were generally influenced less by harvest related regulations. In their review of Smallmouth Bass population characteristics across North America, low productivity was typical of northern and northwestern populations, such as the Pend Oreille River. While system productivity likely influences growth and mortality in the Pend Oreille River Smallmouth Bass population, we reiterate the potential error in our annual mortality estimate due to size bias in electrofishing samples. However, this potential error was addressed in our modeling effort by incorporated a range of natural and fishing mortality rates.

We recommend continuing the existing general bag limit of six fish without a size for Smallmouth Bass in the Pend Oreille River. Based on our modeling effort, existing regulations for Smallmouth Bass provide a balance between opportunity for harvest and provision for quality bass fishing within the potential of the population. Our recommendation is relative to the levels of *cm* and *cf* used in our evaluation. For example, a substantial increase in exploitation might influence the effectiveness of a minimum length limit. Given fishing mortality may change overtime with shifts in angler behavior, we recommend exploitation of Smallmouth Bass in the Pend Oreille River be monitored periodically.

RECOMMENDATIONS

1. Continue periodic monitoring of the Pend Oreille River littoral fish community at five- to ten year intervals.
2. Maintain the existing general bag limit of six fish of any size for Smallmouth Bass in the Pend Oreille River.
3. Periodically monitor exploitation of Smallmouth Bass in the Pend Oreille River.

Table 12. Location, effort, and habitat type of sites sampled during the Pend Oreille River littoral fish community survey in 2016.

Water	Unit	Effort (s)	N	E	Datum	Habitat type
Pend Oreille River	1	600	5336119	501832	WGS84	sand
Pend Oreille River	2	600	5335891	504014	WGS84	sand
Pend Oreille River	3	600	5335854	504780	WGS84	sand
Pend Oreille River	4	600	5335924	507144	WGS84	sand
Pend Oreille River	5	600	5335332	509457	WGS84	sand
Pend Oreille River	6	600	5335215	509759	WGS84	sand
Pend Oreille River	7	600	5334024	510102	WGS84	rock
Pend Oreille River	8	600	5333680	510307	WGS84	rock
Pend Oreille River	9	600	5332914	511562	WGS84	sand
Pend Oreille River	10	600	5332636	511966	WGS84	rock
Pend Oreille River	11	600	5333260	517094	WGS84	rock
Pend Oreille River	12	600	5333778	517582	WGS84	sand
Pend Oreille River	13	600	5334511	519490	WGS84	sand
Pend Oreille River	14	600	5335342	519148	WGS84	rock
Pend Oreille River	15	604	5332864	512610	WGS84	sand
Pend Oreille River	16	609	5332385	512671	WGS84	slough
Pend Oreille River	17	609	5332443	513112	WGS84	slough
Pend Oreille River	18	604	5333411	513740	WGS84	slough
Pend Oreille River	19	604	5332333	514210	WGS84	sand
Pend Oreille River	20	623	5334110	516357	WGS84	sand
Pend Oreille River	21	604	5334466	516692	WGS84	slough
Pend Oreille River	22	621	5333938	516678	WGS84	sand
Pend Oreille River	23	604	5335850	520886	WGS84	slough
Pend Oreille River	24	605	5336869	520708	WGS84	sand
Pend Oreille River	25	619	5337176	520897	WGS84	sand
Pend Oreille River	26	612	5337931	520917	WGS84	rock
Pend Oreille River	27	641	5338418	520919	WGS84	rock
Pend Oreille River	28	603	5338833	520807	WGS84	rock
Pend Oreille River	29	600	5342374	522515	WGS84	slough
Pend Oreille River	30	604	5338669	522636	WGS84	rock
Pend Oreille River	31	600	5342747	523361	WGS84	sand
Pend Oreille River	32	600	5343430	523464	WGS84	sand
Pend Oreille River	33	600	5339608	520885	WGS84	sand
Pend Oreille River	34	600	5340006	521655	WGS84	sand
Pend Oreille River	35	600	5340971	521842	WGS84	rock
Pend Oreille River	36	600	5340610	522592	WGS84	rock
Pend Oreille River	37	600	5340784	522913	WGS84	slough

Table 12 (continued)

Water	Unit	Effort (s)	N	E	Datum	Habitat type
Pend Oreille River	38	600	5342229	522496	WGS84	slough
Pend Oreille River	39	600	5344193	523714	WGS84	sand
Pend Oreille River	40	600	5344901	524091	WGS84	rock
Pend Oreille River	41	600	5343683	526400	WGS84	sand
Pend Oreille River	42	600	5343610	527197	WGS84	sand
Pend Oreille River	43	600	5344308	527138	WGS84	slough
Pend Oreille River	44	600	5344181	527392	WGS84	slough
Pend Oreille River	45	600	5343802	528260	WGS84	slough
Pend Oreille River	46	600	5342618	529289	WGS84	sand
Pend Oreille River	47	600	5342925	531147	WGS84	sand
Pend Oreille River	48	600	5342172	531976	WGS84	sand
Pend Oreille River	49	600	5345010	530766	WGS84	sand
Pend Oreille River	50	600	5345110	531492	WGS84	sand

Table 13. Yield-per-recruit model parameters used to estimate the effect of a 400 mm length limit on abundance of 400 mm Smallmouth Bass in the Pend Oreille River.

Model Parameter	Description	Value
Min TL	minimum harvest length	254 mm/400 mm
N_0	initial population size	1000
b	weight:length function slope	3.239
a	weight:length function intercept	-5.489
Winf (g)	max theoretical weight	1909.061 g
Max Age	max age in the population	9
L_∞	max theoretical length	510 mm
K	growth coefficient	0.182
t_0	theoretical time at TL = 0	-0.862

Table 14. Descriptive statistics for fish species sampled from the Pend Oreille River in June 2016. Statistics summarized include catch, catch rates (fish/h; 80% C.I.) proportion of catch by number and biomass, minimum and maximum total length (TL), and by species.

Species	n	CPUE	% of Catch	% of Biomass	Min of TL	Max of TL
Black Crappie	250	29.9 ± 6.9	12%	6%	75	339
Brook Trout	2	0.2 ± 0.3	> 1%	0%	180	191
Brown Bullhead	56	6.7 ±3.4	3%	6%	202	325
Brown Trout	32	3.8 ±1.1	2%	4%	120	555
Kokanee	8	1.0 ± 0.9	> 1%	0%	48	256
Lake Whitefish	1	0.1 ± 0.2	> 1%	0%	412	412
Largemouth Bass	35	4.2 ± 1.8	2%	5%	65	520
Largescale Sucker	72	8.6 ± 2.0	4%	22%	103	534
Longnose Sucker	2	0.2 ± 0.2	> 1%	0%	337	378
Northern Pikeminnow	47	5.6 ± 2.2	2%	3%	146	560
Peamouth	289	34.6 ± 6.9	14%	8%	96	330
Pumpkinseed	248	29.7 ± 9.9	12%	4%	54	180
Rainbow Trout	11	1.3 ± 0.6	1%	0%	128	311
Smallmouth Bass	307	36.6 ± 8.4	15%	17%	52	480
Tench	36	4.3 ± 3.7	2%	15%	314	485
Walleye	14	1.7 ± 0.7	1%	0%	128	345
Westslope Cutthroat Trout	22	2.6 ± 1.1	1%	1%	155	345
Yellow Perch	610	72.8 ± 14.8	30%	9%	70	265

Table 15. Population metrics for selected warmwater sportfish from Pend Oreille River electrofishing surveys in 2005, 2010, and 2016. Metrics include PSD values, average relative weight (W_r), age at 305 mm, and annual mortality.

Species	Year	PSD ± 95% C.I.	Avg W_r	Age @ 305 mm	Annual Mortality
Black Crappie	2005	2.2 ± 1.0	111	--	84%
Black Crappie	2010	74.4 ± 3.9	--	--	--
Black Crappie	2016	20.5 ± 5.8	100	--	--
Largemouth Bass	2005	37.1 ± 9.2	95	5	34%
Largemouth Bass	2010	93.8 ± 3.9	96	3.7	45%
Largemouth Bass	2016	45.0 ± 21.8	100	--	--
Smallmouth Bass	2005	36.9 ± 11.7	93	4	69%
Smallmouth Bass	2010	36.4 ± 6.2	99	4.6	53%
Smallmouth Bass	2016	20.8 ± 5.5	86	4.2	51%
Yellow Perch	2005	4.0 ± 1.5	87	--	53%
Yellow Perch	2010	12.0 ± 5.3	--	--	--
Yellow Perch	2016	3.5 ± 1.6	79	--	--

Table 16. Results of comparisons of species-specific catch rates from electrofishing surveys of the Pend Oreille River in 2005, 2010, and 2016. Values reported include *P*-values and identified significant differences at $\alpha = 0.20$ by species.

		Significant Differences ($\alpha = 0.20$)		
Species	*P*	2005	2010	2016
Black Crappie	0.00	a	b	b
Brook Trout	0.17	a	b	a
Brown Bullhead	0.18	a	ab	b
Brown Trout	0.00	a	b	a
Kokanee	0.34	a	a	a
Lake Whitefish	0.38	a	a	a
Largemouth Bass	0.01	a	a	b
Largescale Sucker	0.00	a	b	b
Longnose Sucker	0.00	a	b	b
Mountain Whitefish	0.02	a	a	b
Northern Pikeminnow	0.00	a	b	b
Peamouth	0.15	a	b	ab
Pumpkinseed	0.00	a	b	b
Rainbow Trout	0.06	a	a	ab
Redside Shiner	0.00	a	b	b
Smallmouth Bass	0.37	a	a	a
Tench	0.56	a	a	a
Walleye	0.03	a	b	ab
Westslope Cutthroat Trout	0.72	a	a	a
Yellow Perch	0.00	a	b	a

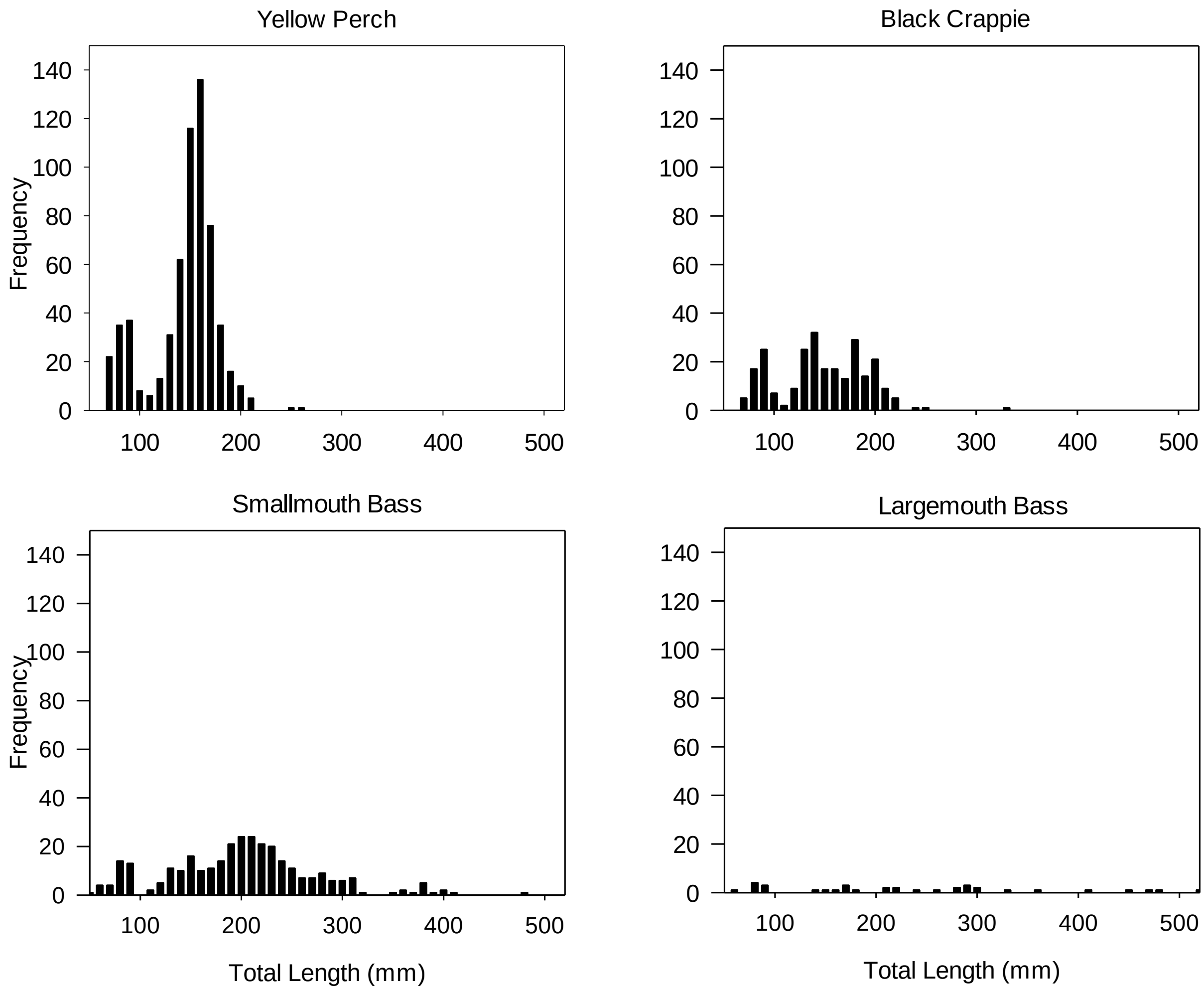

Figure 19. Length-frequency histograms of Yellow Perch, Black Crappie, Smallmouth Bass, and Largemouth Bass sampled from the Pend Oreille River in 2016.

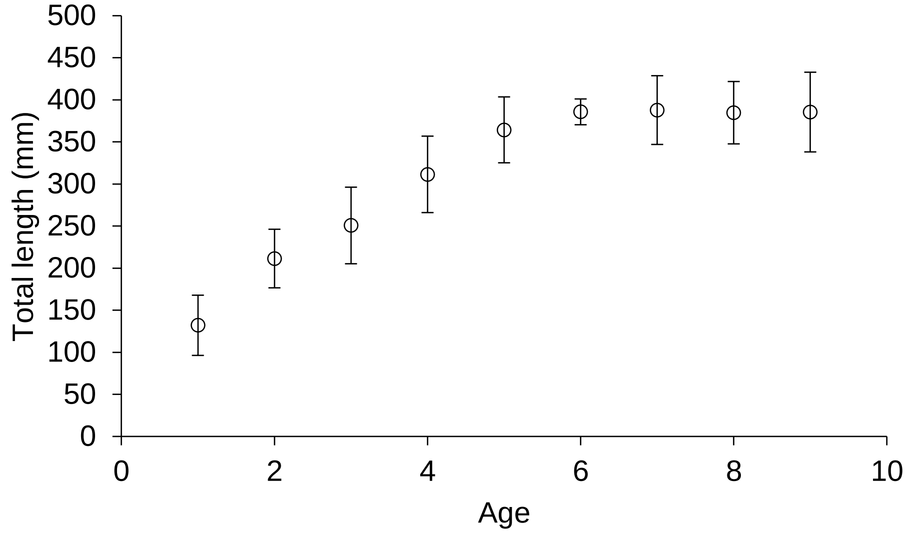

Figure 20. Mean total length-at-age (± 1 SD) at time of sampling for Smallmouth Bass collected from the Pend Oreille River in 2016.

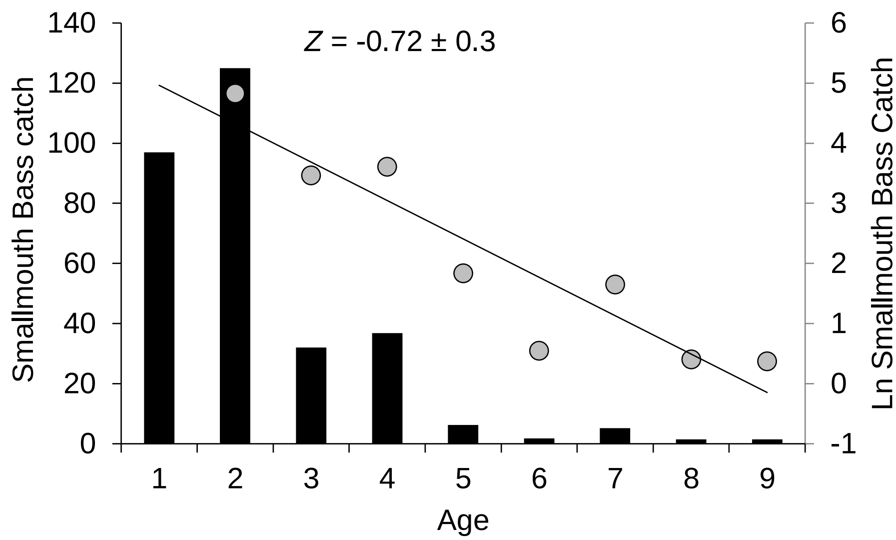

Figure 21. Catch-at-age and associated catch curve used to estimate instantaneous natural mortality (± 80% C.I.) from Pend Oreille River Smallmouth Bass collected in 2016.

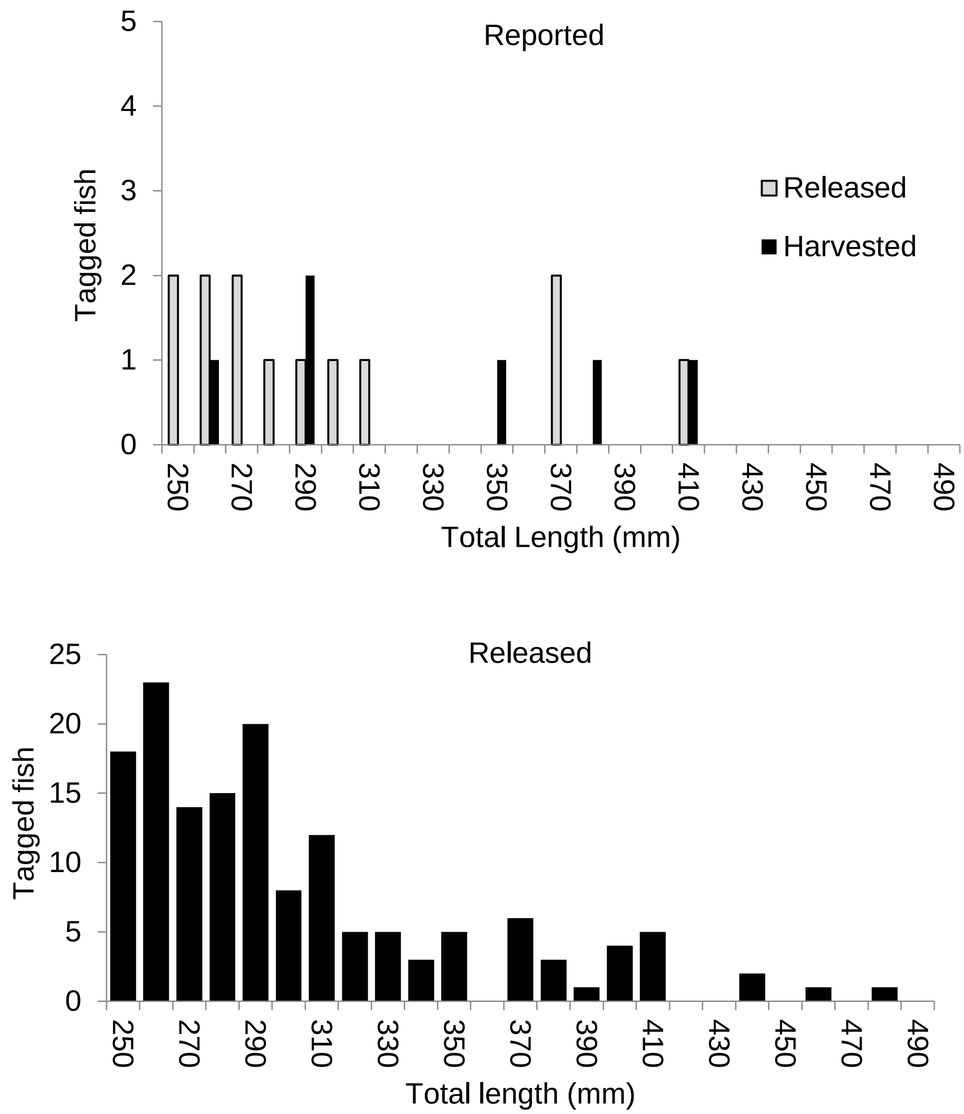

Figure 22. Length-frequency of Smallmouth Bass tagged with T-bar tags and released in the Pend Oreille River in 2016 (bottom panel) and length-frequency of fish caught and reported by anglers from June 2016 through December 2016 (top panel). Angler caught fish represented both total caught and that portion caught and released.

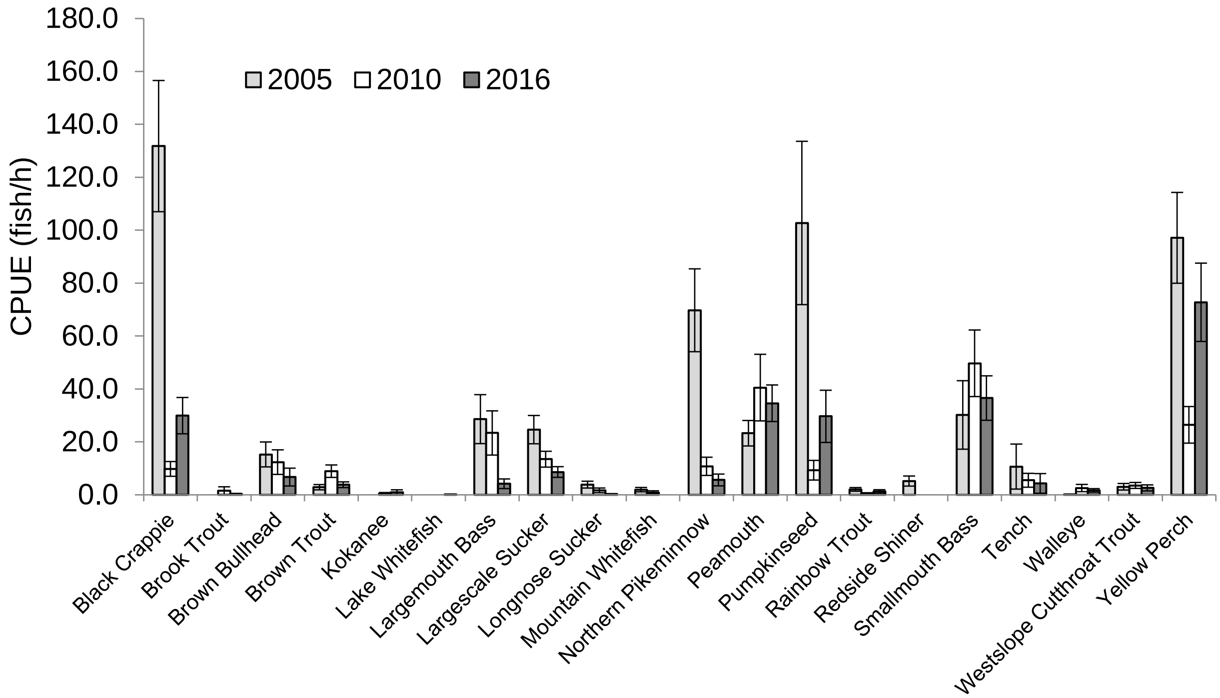

Figure 23. Mean CPUE estimates and 80% confidence intervals for all species sampled in littoral electrofishing surveys of the Pend Oreille River from 2005, 2010, and 2016.

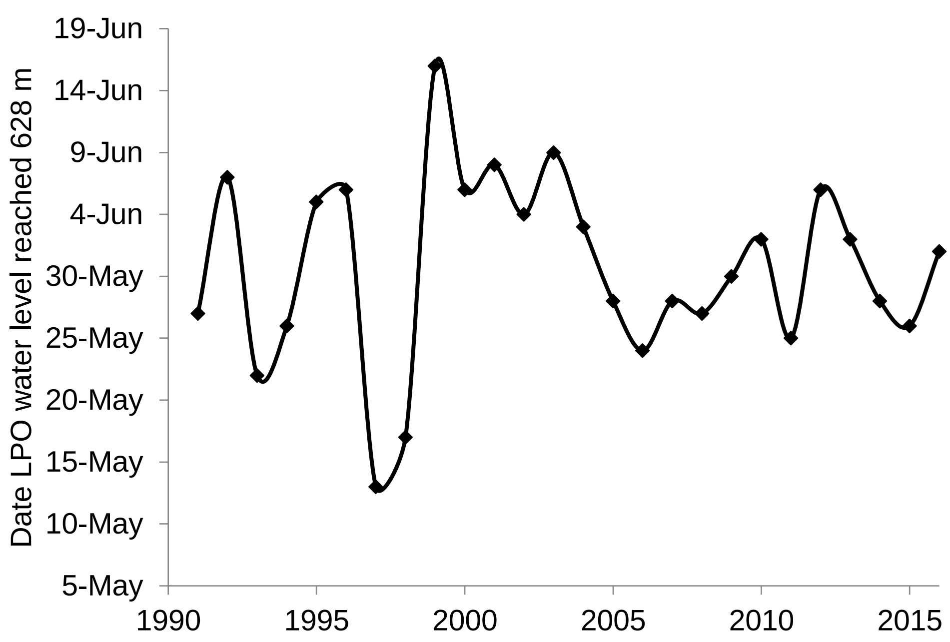

Figure 24. Month and day by year at which elevation of Lake Pend Oreille reached 628 m.

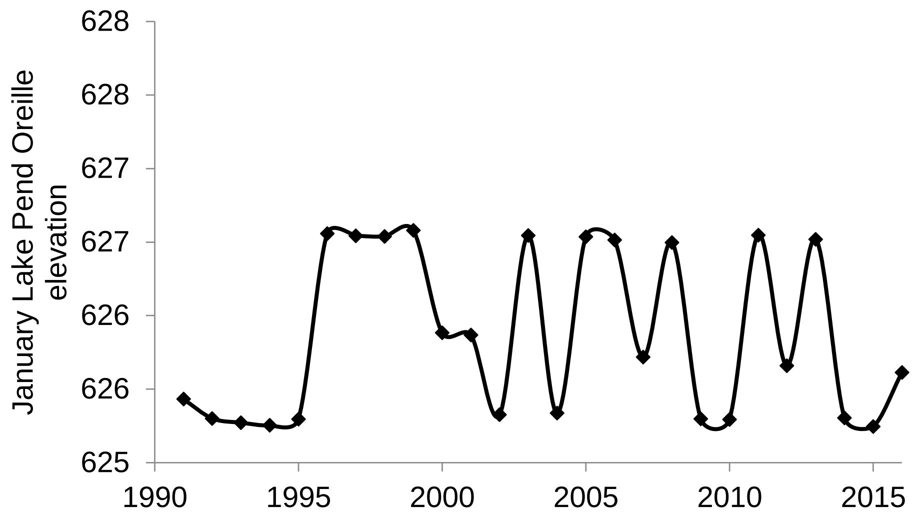

Figure 25. Mean January surface elevation of Lake Pend Oreille by year.

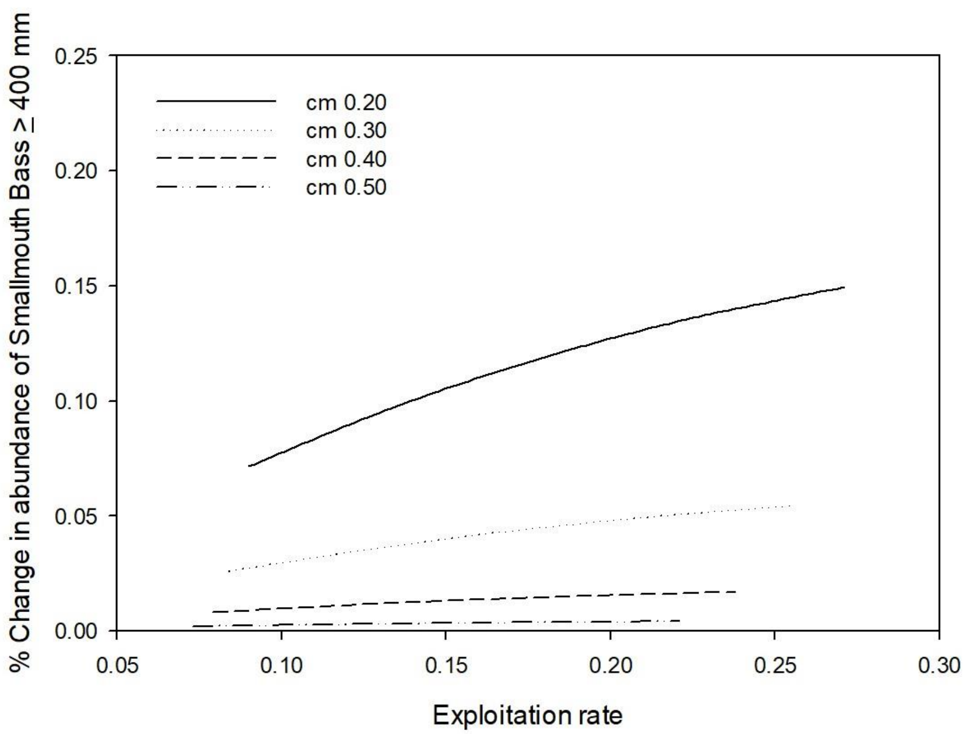

Figure 26. Predicted differences in the proportion of a Smallmouth Bass cohort reaching 400 mm between two modeled regulations on the Pend Oreille River. Regulations included no length limit and a 400 mm minimum length limit. Regulations were modeled over a range of conditional natural mortality rates (cm) and exploitation rates.

PEND OREILLE RIVER TRIBUTARY INVENTORIES

ABSTRACT

The Pend Oreille River originates at the outflow of Lake Pend Oreille in northern Idaho. It flows west for 42 km through the Idaho Panhandle, north through northeastern Washington into British Columbia, and west to its confluence with the Columbia River. The fish community of the Idaho segment of the Pend Oreille River is diverse, including both a variety of native and non-native fishes. Tributaries to the Pend Oreille River are thought to provide spawning and rearing habitat for migratory salmonids and support resident fish populations. However, little information is available on many of the fish populations occurring in these streams. We surveyed first- and second-order tributaries of the Pend Oreille River in 2018 to describe fish species presence, abundance, and distribution. Water was present and an electrofishing survey was completed at 20 of the 21 sites visited. Brook Trout *Salvelinus fontinalis* and Westslope Cutthroat Trout *Onchorhynchus clarkii* were common among most streams and represented the only salmonid species collected. Westslope Cutthroat Trout were found in 8 of the 10 tributaries surveyed. Mean densities varied from 0.4 to 39.8 fish/100 m^2. Brook Trout were also widely distributed, being found in seven of the 10 tributaries surveyed. Mean densities varied from 0.7 to 80.4 fish/100 m^2. Our survey suggested Westslope Cutthroat Trout remain well-distributed throughout tributaries of the Pend Oreille River.

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INTRODUCTION

The Pend Oreille River originates at the outflow of Lake Pend Oreille in northern Idaho. It flows west through the Idaho Panhandle, north through northeastern Washington into British Columbia, and west to its confluence with the Columbia River. Approximately 42 km of the river occur within Idaho. Albeni Falls Dam, located on the Pend Oreille River near the border of Idaho and Washington, regulates water levels in the river and Lake Pend Oreille. River and lake water elevation seasonally fluctuates up to 3.5 m between summer full pool and winter drawdown.

The fish community of the Pend Oreille River is diverse, including both a variety of native and non-native fishes. Species present include Black Crappie *Pomoxis negromaculatus*, Brook Trout *Salvelinus fontinalis*, Brown Bullhead *Ameiurus nebulosus*, Brown Trout *Salmo trutta*, kokanee *Oncorhynchus nerka*, Lake Whitefish *Coregonus clupeaformis*, Largemouth Bass *Micropterus salmoides*, Largescale Sucker *Catostomus macrocheilus*, Longnose Sucker *Catostomus catostomus*, Northern Pikeminnow *Ptychocheilus oregonensis* , Peamouth *Mylocheilus caurinus*, Pumpkinseed *Lepomis gibbosus*, Rainbow Trout *Oncorhynchus mykiss*, Smallmouth Bass *Micropterus dolomieu*, Tench *Tinca tinca*, Walleye *Stizostedion vitreum*, Westslope Cutthroat Trout *Oncorhynchus clarkii*, and Yellow Perch *Perca flavescens* (Bennett and Dupont 1993, Maiolie et al. 2011). Redside Shiner *Richardsonius balteatus*, once common in the system, are now rare (Maiolie et al. 2011). Species composition and abundance in the Pend Oreille River are thought to be heavily influenced by water level management (Bennett and Dupont 1993, Schoby et al. 2007, Maiolie et al. 2011). Unintended introduction of new fish species (i.e., Smallmouth Bass, Walleye) has also influenced composition and abundance within the fish community (Schoby et al. 2007, Maiolie et al. 2011).

Tributaries to the Pend Oreille River are thought to provide spawning and rearing habitat for migratory salmonids and support resident fish populations. However, little information is available on many of the fish populations occurring in these streams. The Idaho Department of Fish and Game 2019-2024 Fisheries Management Plan identifies multiple objectives which highlight a need for knowledge of fish communities in Pend Oreille River tributaries (IDFG 2013). To address these objectives, we surveyed first and second order tributaries of the Pend Oreille River in 2018 to describe fish species presence, abundance, and distribution.

METHODS

We visited 21 sites among 10 tributaries to the Pend Oreille River in June and July of 2018 (Table 17). Streams sampled included Carey, Carr, Fry, Hornby, Johnson, Moore, Riley, Smith, and Syringa creeks, as well as an unnamed tributary. Sampled tributaries were selected to provide a largescale view of fish distribution and abundance throughout the basin. In an effort to describe species distribution and abundance, sampling followed a systematic design within each tributary. Individual sample sites were distributed from a stream's confluence with the Pend Oreille River to headwater reaches of each tributary. Although sample sites were distributed throughout each tributary, the uppermost sample sites in most cases did not represent the full distribution of fish habitat. We sampled one to three sites per stream at pre-determined locations. Locations were identified *a priori* using ArcMap (Environmental Systems Research Institute). Access to most tributaries was difficult because much of the surrounding land ownership is private. Many landowners granted permission to access selected sites. However, access was denied or no contact with a landowner was made to request access at some locations. If possible, we relocated sample sites when access was not found to a selected site, but was available nearby.

We collected fish using a Smith-Root LR-24 backpack electrofisher. Settings varied among sites due to differences in water conductivity, but generally included 60Hz and 700 to 800 V. Most sites were sampled by two people, with one person shocking and another netting fish. Sample sections were typically 100 m in length. We closed sample sections using a block net at the downstream end of a survey section at most sites to prevent escapement during downstream electrofishing passes. On multi-pass samples, we completed sequential passes until captures of an individual pass were no more than 20% of the total capture by species summed across all passes. Typically, two or three passes were completed. All fish collected were identified, measured (total length, mm) and released downstream of the sample transect. We estimated the width of each sample transect using an average of multiple width measurements. Measured widths were spaced at 10 - 20 m intervals throughout each transect.

Abundance of tributary fish populations was estimated using multi-pass removal estimates (Zippin 1958) in combination with single-pass samples. We derived abundance estimates and associated 80% confidence intervals for two- and three-pass samples using calculations for removal estimates in closed populations (Hayes et. al 2007). We reported the total catch on the first pass as the population estimate when all the individuals of a particular species were captured on the first pass. In cases where lower confidence bounds were less than the total number of fish captured, the total number of fish captured was reported as the lower bound. Single-pass sampling was used to increase the number of possible sample sites surveyed. We estimated abundance from single-pass samples by generating a multi-pass regression model of abundance based on first-pass collections (Meyer and Schill 1999). A single model of abundance based on first-pass collections was developed and included sample data from all tributaries and all target species. Capture efficiencies were consistent among all tributaries and species providing support that model predictions were valid across these boundaries. Abundance estimates included fish ≥75 mm total length due to low sampling efficiency on smaller fish. We reported density estimates as the number per 100 m^2. We also used sampled fishes to describe population characteristics within sampled streams, such as size structure and species composition.

RESULTS

Water was present and an electrofishing survey was completed at 20 of the 21 sites visited (Table 17). Water was limited and no fish were detected at the single site visited on Carey Creek. Fish were detected in all other streams (Table 18).

A single regression model was developed to estimate abundance based on first-pass collections (Figure 27). Capture efficiency in multi-pass samples was consistent (0.67 ± 0.13, 1 SD.) among tributaries and species, providing support that our model predictions were valid across these variables. Based on the developed linear model, our first-pass collections described approximately 95% of the variation in estimated abundance from multi-pass samples.

We found fish communities in surveyed tributaries were simple with few represented species. Brook Trout and Westslope Trout were common among most streams and represented the only salmonid species collected. Sculpin *Cottus spp*. were detected in Johnson, Riley, and Syringa creeks. Black Bullhead were also caught, but only in Hornby Creek. Small fish (< 75 mm), assumed to be age-0, were caught at several locations. Although collectors were able to identify these fish as *Oncorhynchus spp.*, they were not able to positively identify to the species level. Rainbow Trout were not detected at any site, suggesting these fish were Westslope Cutthroat Trout. Small Brook Trout were generally identifiable, regardless of size.

Westslope Cutthroat Trout were found in 8 of the 10 tributaries surveyed (Table 19). Mean densities varied from 0.4 to 39.8 fish/100 m^2 (Table 18). Westslope Cutthroat Trout were not detected in Smith Creek. Total length of sampled Westslope Cutthroat Trout varied from 27 to 257 mm (Table 19).

Brook Trout were also widely distributed, being found in 7 of the 10 tributaries surveyed (Table 19). Mean densities varied from 0.7 to 80.4 fish/100 m^2 (Table 18). Brook Trout were not detected in Moore Creek or Unnamed Creek. Total length of sampled Brook Trout varied from 31 to 261 mm (Table 19).

DISCUSSION

Westslope Cutthroat Trout remain well-distributed throughout tributaries of the Pend Oreille River. We found densities representing low to high abundance relative to other populations in the region (Ryan et al. 2020b, Bouwens et al. 2019). Although we found Westslope Cutthroat Trout were widely distributed, their abundance may be reduced as a result of competition with Brook Trout which were also well distributed.

Information obtained from this inventory established baseline knowledge for fish populations in the sampled streams. Although, a description of trends in fish communities was not possible, this information did improve knowledge of where fish species occur in the drainage and where conservation priorities may exist. We recommend the information obtained in this inventory be applied to future actions aimed at conserving native fish in the Pend Oreille drainage, including both resident and migratory populations. Furthermore, both resident and migratory life history types may exist in the surveyed streams. This inventory did not describe the life history types of fish species encountered. However, this work did provide guidance to direct future investigations aimed at defining the life history strategies of native fish and, specifically, the origin of migratory Westslope Cutthroat Trout in the drainage.

Brook Trout were dominant in five of the seven streams in which they were detected. Densities represented low to very high abundance relative to other populations in the region. In comparison, mean stream-wide density of Brook Trout in tributaries of Priest/Upper Priest Lakes, Lake Pend Oreille, and the Kootenai River varied from 0.3 fish/100 m^2 to 16.5 fish/100 m^2 (Ryan et al. 2020b, Bouwens et al. 2019, see Kootenai River Redband Trout Inventory in this report). Brook Trout density in Syringa Creek (80.4 fish/100 m^2) was more similar to densities described in sink drainages of the Rathdrum Prairie which varied from 10.4 fish/100 m^2 to 117.1 fish/100 m^2 (Ryan et al. 2014). Although Brook Trout were historically introduced widely throughout the region, other regional investigations have noted their expansion and dominance in lower gradient, lower velocity and or altered systems (Griffith 1972; Ryan et al. 2020b). We did not measure channel slope or other metrics of habitat quality. However, we did generally observe many of the surveyed stream reaches represented both low to moderate gradient and altered condition habitats. As previously noted, the lands surrounding surveyed streams largely consisted of private lands. Land uses varied and included agriculture, timber, and rural residential development. We saw related conditions such as channelization, water diversion, and sedimentation that reduced habitat quality. We hypothesized the dominance of Brook Trout was likely due in part to the existing habitat.

Water quantity may influence fish abundance in some tributaries to the Pend Oreille River. In lower Carey Creek, we found little flowing water and an absence of fish in late-June. In Fry Creek, we found water, but quantity was limited. At that site, a high density of *Oncorhynchus* fry were caught, suggesting fish may be concentrated where water was present. Water diversion from Carr Creek (to Hornby Creek) was observed and may influence water flow seasonally, although it was not apparent at the time of our survey. Fish abundance in Carr Creek was low relative to other tributaries, suggesting some limitation exists. The lowermost site on Unnamed Creek also had few fish. Adjacent landowners to this site suggested flow was often low or absent in late-summer. The majority of these drainages are small, low elevation basins. Because of their size and location, they may be sensitive to annual fluctuations in snow fall. Water use, as in the case of Carr Creek, likely exacerbates these issues. We recommend highlighting the value of high quality water and land resources with private landowners in this area as a method of encouraging habitat conservation for fish populations.

RECOMMENDATIONS

1. Apply information obtained in this inventory to future actions aimed at conserving native fish in the Pend Oreille drainage, including both resident and migratory populations.

2. Highlight the value of high quality water and land resources in the Pend Oreille River drainage with private landowners as a method of encouraging habitat conservation for fish populations.

Table 17. Locations of sites sampled during 2018 surveys of Pend Oreille River tributaries. Water temperature, site length and average wetted width at the time of sampling are listed for each survey site.

Stream	Site	Latitude	Longitude	Date	Temp (C°)	Length (m)	Avg width (m)
Carey Creek	Carey 1	48.1408723	-116.846269	6/25/2018	--	--	--
Carr Creek	Carr 1	48.2696200	-116.666779	7/11/2018	11.0	105.0	4.5
	Carr 2	48.2856587	-116.661369	7/10/2018	11.0	107.0	3.4
	Carr 3	48.300586	-116.663597	6/26/2018	12.0	96.0	2.3
Fry Creek	Fry 1	48.194816	-116.532354	7/12/2018	13.5	31.7	2.1
Hornby Creek	Hornby 1a	48.258958	-116.628841	7/9/2018	19.0	101.3	3.0
	Hornby 2	48.269124	-116.638096	7/11/2018	18.0	104.5	2.1
	Hornby 3	48.279875	-116.646977	7/10/2018	17.5	94.0	2.2
Johnson Creek	Johnson 2	48.236060	-116.714850	7/3/2018	11.0	109.5	2.5
	Johnson 3	48.247239	-116.727298	7/3/2018	9.0	110.0	1.5
Moore Creek	Moore 2	48.181437	-116.668522	7/12/2018	16.0	108.5	1.4
Riley Creek	Riley 1	48.174658	-116.762490	6/27/2018	13.0	97.0	3.3
	Riley 2a	48.194368	-116.754159	6/27/2018	--	90.0	3.5
	Riley 3	48.214932	-116.764354	6/28/2018	--	101.7	3.9
Smith Creek	Smith 2	48.269071	-116.711333	7/2/2018	11.0	114.0	1.7
	Smith 3	48.280345	-116.711746	7/2/2018	9.0	110.0	1.2
Syringa Creek	Syringa 2	48.288501	-116.587450	7/12/2018	15.0	44.5	2.2
	Syringa 3	48.296235	-116.600849	7/11/2018	13.0	101.3	1.9
Unnamed Creek	Unnamed 1	48.165049	-116.905062	7/11/2018	17.5	106.0	1.4
	Unnamed 2	48.151726	-116.906423	7/11/2018	16.0	109.0	1.1
	Unnamed 3	48.141096	-116.912733	6/25/2018	--	100.0	1.9

Table 18. Mean density of salmonids in surveyed tributaries to the Pend Oreille River in 2018. Density estimates represent only fish ≥ 75 mm. Mean density values were calculated by species for all surveyed sections per stream.

Stream	Brook Trout (#/100 m^2)	Westslope Cutthroat Trout (#/100 m^2)
Carr Creek	0.7	4.0
Fry Creek	1.9	5.6
Hornby Creek	8.3	0.4
Johnson Creek	12.3	5.6
Moore Creek	--	1.7
Riley Creek	11.7	2.8
Smith Creek	12.3	--
Syringa Creek	80.4	29.4
Unnamed Creek	--	39.8

Table 19. Pend Oreille River tributary 2018 survey results by stream, sampled section, and species. Catch and length distributions (mean and maximum total length – TL) includes fish of all lengths (mm), while only fish ≥ 75 mm were included in abundance estimates (Est. N).

Stream	Site	Species	Catch	Mean TL(SD)	Max TL	Est. N	80% CI-	80% CI +	Fish/100m^2
Carr Creek	1	BRK	10	51 (6)	59	--	--	--	--
Carr Creek	1	WCT	11	101 (20)	143	12.6	10	18.2	2.7
Carr Creek	2	BRK	2	101 (5)	104	2.5	2	8.2	0.7
Carr Creek	2	WCT	16	122 (22)	221	19	15	24.5	5.2
Fry Creek	1	BRK	1	187	187	1.3	1	6.9	1.9
Fry Creek	1	Onc Fry	279	46 (8)	64	--	--	--	--
Fry Creek	1	WCT	5	125 (4)	130	3.8	3	9.4	5.6
Hornby Creek	1a	BBH	18	119 (13)	136	--	--	--	--
Hornby Creek	1a	BRK	24	81 (20)	125	15.2	12	20.7	5
Hornby Creek	1a	WCT	1	142	142	1.3	1	6.9	0.4
Hornby Creek	2	BBH	23	101 (17)	131	--	--	--	--
Hornby Creek	2	BRK	81	79 (35)	200	27.6	25	31.3	12.5
Hornby Creek	3	BBH	2	81	81	--	--	--	--
Hornby Creek	3	BRK	22	92 (43)	182	15.2	12	20.7	7.3
Johnson Creek	2	BRK	140	86 (44)	202	62.7	61	64.9	23.1
Johnson Creek	2	Onc Fry	22	33 (4)	40	--	--	--	--
Johnson Creek	2	SCP	88	53 (14)	92	--	--	--	--
Johnson Creek	2	WCT	25	109 (20)	160	26.1	25	28	9.6
Johnson Creek	3	BRK	19	59 (32)	170	2.5	2	8.2	1.5
Johnson Creek	3	Onc Fry	7	28 (4)	33	--	--	--	--
Johnson Creek	3	WCT	2	112 (33)	135	2.5	2	8.2	1.5
Moore Creek	2	WCT	2	211 (65)	257	2.5	2	8.2	1.7
Riley Creek	1	BRK	7	114 (75)	219	3.8	3	9.4	1.2
Riley Creek	1	SCP	7	84 (16)	98	--	--	--	--

Table 19 (continued)

Stream	Site	Species	Catch	Mean TL(SD)	Max TL	Est. N	80% CI-	80% CI +	Fish/100m^2
Riley Creek	1	WCT	6	90 (45)	123	5.1	4	10.7	1.6
Riley Creek	2a	BRK	154	84 (43)	210	67	61.5	72.5	21.4
Riley Creek	2a	SCP	59	57 (13)	96	--	--	--	--
Riley Creek	2a	WCT	2	186 (1)	186	2.5	2	8.2	0.8
Riley Creek	3	BRK	70	96 (38)	177	49.4	48	51.5	12.4
Riley Creek	3	Onc Fry	15	27 (1)	29	--	--	--	--
Riley Creek	3	SCP	287	53 (14)	166	--	--	--	--
Riley Creek	3	WCT	24	89 (29)	158	24	21	28.6	6
Smith Creek	2	BRK	27	99 (49)	226	22.8	18	28.3	11.9
Smith Creek	3	BRK	26	104 (57)	182	16.3	16	17.3	12.6
Syringa Creek	2	BRK	83	140 (64)	261	79.6	74	85.3	80.4
Syringa Creek	2	SCP	5	65 (21)	88	--	--	--	--
Syringa Creek	3	WCT	50	103 (27)	167	55.6	50.1	61.1	29.4
Unnamed Creek	1	WCT	1	157	157	1.3	1	6.9	0.8
Unnamed Creek	2	Onc Fry	39	32 (4)	38	--	--	--	--
Unnamed Creek	2	WCT	136	98 (28)	207	118.6	116	121.5	98.1
Unnamed Creek	3	WCT	32	100 (22)	157	37.9	32.4	43.4	20.4

BBH = Black Bullhead
BRK = Brook Trout
Onc Fry = Unidentified Oncorhynchus Fry (age-0)
SCP = Sculpin Species
WCT = Westslope Cutthroat Trout

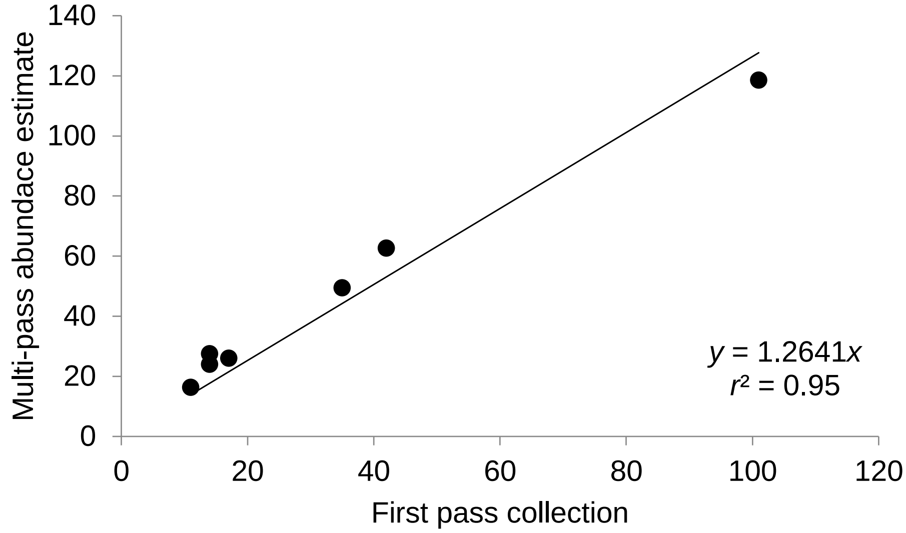

Figure 27. Linear model predicting the relationship between multi-pass abundance estimates and first-pass catch from Pend Oreille River tributaries sampled in 2018.

COCOLALLA LAKE INVESTIGATIONS 2018

ABSTRACT

A survey of the Cocolalla Lake fish community was completed in May and June 2018. Survey methods followed Idaho Department of Fish and Game lowland lake standard protocol. A year-long angler survey was also completed on Cocolalla Lake from April 2018 through March 2019. We collected 15 species during our lowland lake survey. Yellow Perch *Perca flavescens* and Channel Catfish *Ictalurus punctatus* were the most abundant species sampled. Rainbow Trout *Oncorhynchus mykiss* caught in our sample were representative of recently stocked catchable size fish. Abundance, size structure, and condition of most species was similar to prior surveys of the fish community. However, population-level changes in Black Crappie *Pomoxis nigromaculatus*, Channel Catfish, Largemouth Bass *Micropterus salmoides*, and Yellow Perch were detected. Anglers fished an estimated 19,733 hours and reported catching 11 species during the surveyed period. Collectively, salmonids were the most targeted group of fish. However, a majority of anglers fishing Cocolalla Lake were generalists and did not specifically target any one species. Catch rates varied widely by species and season.

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Regional Fishery Biologist

INTRODUCTION

Cocolalla Lake is located in Bonner County, Idaho, near the community of Westmond. The lake's surface area is approximately 325 hectares. Maximum depth is approximately 12 m. The lands surrounding the lake are primarily private ownership and many residences are located near the lakeshore. An Idaho Department of Fish and Game (IDFG) access site on the north side of the lake provides the only public boating access to the lake. Water depth at the IDFG boat ramp can limit access for larger boats from mid-summer through fall. An IDFG wildlife management area property abuts the lake on its southern end, but access to the lake through the property is undeveloped walk-in access only.

Cocolalla Lake is managed as a mixed species fishery under general regional bag and size limits. Rainbow Trout *Oncorhynchus mykiss* were stocked historically in the lake at both catchable length (i.e., 152 - 305 mm) and fingerling length (i.e., 76 – 152 mm). Fingerling Rainbow Trout were stocked at moderate densities (e.g., 77 fish/ha) from 2011 through 2016, but provided little return to the fishery (Ryan et al. 2020b). Stocking of catchable Rainbow Trout was reinitiated in 2018 with an interest in improving Rainbow Trout fishing. Fingerling Westslope Cutthroat Trout *Oncorhynchus clarkii* have also been stocked periodically over time at moderate densities (e.g., 62 fish/ha) and were present during surveys of the lake, suggesting they provided fishing opportunity (Ryan et al. 2020b). Catchable Channel Catfish *Ictalurus punctatus* have been stocked at varying rates and frequencies in Cocolalla Lake since 1985. Brook Trout *Salvelinus fontinalis* and Brown Trout *Salmo trutta* were historically stocked in the drainage and now persist through natural recruitment (Davis et al. 1996, Fredericks et al. 2009). Warmwater species previously identified in Cocolalla Lake include Black Crappie *Pomoxis nigromaculatus,* Bluegill *Lepomis macrochirus,* Brown Bullhead *Ameiurus nebulosus,* Largemouth Bass *Micropterus salmoides*, Pumpkinseed *Lepomis gibbosus,* and Yellow Perch *Perca flavescens* (Davis et al. 1996, Fredericks et al. 2009). Non-game fishes, including Peamouth *Mylocheilus caurinus*, Largescale Sucker *Catostomus macrocheilus*, Longnose Sucker *C. catostomus*, and Bridgelip Sucker *C. columbianus* have also been observed in previous surveys of the lake (Davis et al. 1996, Fredericks et al. 2009).

Lowland lakes provide a diversity of angling opportunities in the Idaho Panhandle Region. Lowland lake surveys are conducted periodically to monitor the composition and quality of these fisheries. Many lowland lakes within the Panhandle Region are routinely stocked to enhance fishing opportunities. Lowland lake surveys also provide a means of evaluating hatchery stocking for enhancement of lowland lake fisheries. Similarly, angler surveys are conducted periodically to evaluate fishery use and performance. These surveys also provide opportunity to receive feedback from anglers about the fishery.

In 2018, we conducted a lowland lake survey on Cocolalla Lake to describe the current fish community. We used information collected to evaluate the current status of the fishery and performance of hatchery products used in the lake. We also completed a year-long angler survey to describe fishery use, quality, and angler preferences.

METHODS

Lowland Lake Survey

We conducted a lowland lake survey on Cocolalla Lake from May 29 to June 5, 2018. The survey was conducted using Idaho Department of Fish and Game (IDFG) lowland lake standard methods (IDFG 2012). Survey effort included 10 trap net-nights, 10 gill net-nights (5 floating and 5 sinking standard experimental gillnets), and 9 electrofishing units (600 s; Table 20).

Fish collected during surveys were identified, measured (total length, mm) and weighed (g). We estimated relative abundance from electrofishing and netting samples as catch-per-unit-effort (CPUE) standardized to fish per hour and fish per net, respectively. We described the general structure of the fish community as the relative percentage of each species in the sample by count and biomass. Size structure of sampled species was described using length-frequency histograms and by proportional stock densities (PSD; Anderson and Neumann 1996) for primary species targeted. We used Fisheries Analysis and Modeling Simulator (FAMS, Slipke and Maceina 2014) to calculate PSD values. Average relative weight (W_r, Wege and Anderson 1978, Richter 2007) was used to describe the condition of fish.

Hard structures were collected from a subsample of targeted species caught during our survey to describe a length-at-age relationship. We collected dorsal spines from Largemouth Bass and otoliths from a sample of Yellow Perch. We targeted three to five structures per centimeter length group for each species. Dorsal spines were mounted in epoxy, cross-sectioned on a Buehler Isomet saw (Illinois Tool Works Inc., Lake Bluff, Illinois), sanded for viewing clarity, and viewed on a compound microscope under 10x to 30x magnification. Otoliths were broken centrally on the transverse plane, browned, sanded on the broken surface, and viewed under a dissecting microscope using a fiber optic light to illuminate the broken surface. Length-at-age at time of capture was reported as an index of growth where applicable. Age-length-keys were used to predict ages for an entire sample using subsampled age estimates. Length-at-age information was used to describe patterns of growth, mortality, and recruitment. We used a frequency of catch by age for sampled fish in describing general patterns of recruitment and in estimating annual mortality. Annual mortality was estimated by applying a weighted catch curve generated in FAMS.

Trends in Cocolalla Lake species composition and population characteristics were described by comparing metrics from this survey to previously completed surveys. Lowland lake surveys were previously completed on Cocolalla Lake in 1992 and 2008 (Davis et al. 1996, Fredericks et al. 2009). Relative abundance (CPUE) compared among surveys included only electrofishing data. Analysis of netting data differed among surveys, making direct comparisons difficult. Survey timing also differed among surveys. In 1992, the survey was conducted twice, including events in March and early-July. Only the July survey was included in these comparisons. In 2008, the survey occurred in mid-May.

Angler Survey

We conducted a year-long angler survey on Cocolalla Lake from April 2018 through March 2019 using a roving-access design (Pollock et al. 1994). Survey design and analysis was completed using a customized creel survey database (Josh McCormick, IDFG, personal communication). The survey period was divided into two-week intervals in April through September. One-month intervals were used in October through March. Intervals were stratified by day type, including weekdays and weekend/holidays. We scheduled four survey days per interval, including two weekdays and two weekend/holidays. Survey dates were randomly chosen. Daily start times for an eight hour survey shift were also randomly chosen. We coordinated the Cocolalla Lake survey with a concurrent survey of Spirit Lake. Eight-hour shifts were divided into two four-hour periods, one period per fishery. The first period was alternated between fisheries.

Roving counts of boats and shore anglers were conducted twice per shift at randomly scheduled times to estimate angler effort. Creel clerks made a single loop around the lake by boat for each scheduled count. During periods when the lake was iced covered, anglers were counted at the primary IDFG access site and from Highway 95 running parallel to the lake. Few anglers were encountered during ice covered periods at locations other than the primary IDFG access site. Ice anglers were identified as shore anglers for the purpose of survey analysis.

Angler interviews were conducted to obtain catch rate information and describe angler type. Interviews were completed at the IDFG access site on the north end of the lake. Creel clerks waited at the access site to intercept anglers leaving the lake upon completion of their angling effort. We attempted to interview all angling parties leaving through the access site during the survey period. Interview questions included number of anglers, angler type (boat or shore), number of rods fished, time spent fishing, targeted species, number of fish kept per species, number of fish released per species, and whether a daily trip was completed.

Daily fishing effort was first estimated for each day within a sampling interval for which a survey was completed. Daily fishing effort was estimated as average angler count within a sampling interval multiplied by the number of possible fishing hours in the sampled day. Fishing hours were described as the period between sunset and sunrise. Daily fishing effort was expanded to the temporal strata (i.e., two-week period or month and day type) by dividing by the sampling probability:

$$E = e/pt,$$

where E = total effort, e = sampling period effort (daily fishing effort), and pt = temporal sampling probability. Sampling probabilities were estimated by day type as the number days sampled within a strata divided by the number of days within the strata. Fishing effort estimates by day type were then summed across strata for an estimate of total fishing effort by month.

Catch rate was reported as the number of fish caught, harvested, or released per angler hour. Catch rate was estimated from completed trip interviews and was calculated by the ratio of means estimator (Pollock et al. 1994). Total catch was divided by total angler effort for various hierarchies of the survey design (i.e., monthly or total). The total number of fish released, harvested, and caught (harvest + release) were estimated by multiplying total fishing effort by the appropriate total rate estimator (harvest, release, or catch) for the various hierarchies of the design. Total catch was estimated as the product of catch rate and effort for each strata.

Survey metrics (e.g., effort, catch rate, catch) were compared with a prior angler survey conducted in 1992 (Davis et al. 1996) to describe general changes in the fishery. The 1992 survey included only the period from April to September. As such, surveys were not directly comparable, but provided a coarse evaluation of trends.

RESULTS

Lowland Lake Survey

We sampled 15 species from Cocolalla Lake including Black Crappie, Brook Trout, Brown Bullhead, Brown Trout, Channel Catfish, Largemouth Bass, Largescale Sucker, Longnose Sucker, Peamouth, Pumpkinseed, Rainbow Trout (hatchery and wild origin), Rainbow x Westslope Cutthroat Trout hybrids, Smallmouth Bass *Micropterus dolomieu*, Westslope Cutthroat

Trout, and Yellow Perch (Table 21). Yellow Perch were the most abundant species sampled, comprising 26% of the catch by number and 4% of the biomass. Electrofishing was the most effective method of capture for sampling Yellow Perch (128 fish/hour; ± 77, 1 SD; Table 22). Channel Catfish were also abundant, comprising 20% of the catch and 41% of the biomass. Sinking gill nets most effectively captured Channel Catfish (27.4 fish/net ± 5.7; Table 22). Collectively, salmonids of all species represented approximately 13% of the catch. Hatchery Rainbow Trout were the most common salmonid at 9% of the total catch. Smallmouth and Largemouth bass represented 7% and 4% of the catch, respectively. Electrofishing was the most effective capture method for bass with catch rates of 28.6 and 16 fish/hour for Largemouth and Smallmouth bass, respectively (Table 22). Black Crappie were well-represented in our catch, comprising 8% by number and 5% of the biomass. Sinking gill nets had the highest catch rates of Black Crappie at 17.4 fish/net (Table 22).

Small Yellow Perch dominated our catch. Total length varied from 44 to 253 mm and was represented by a PSD of 15 (Table 21, Figure 28). Although age classes from 1-8 were represented in our sample, age classes 1-4 dominated the population (Figure 29). We estimated Yellow Perch grew to 254 mm in 6.4 years (Figure 30). Estimated annual mortality from age-2 to age-8 was 60% (Figure 31). Yellow Perch exhibited average condition (W_r = 97).

Total length of Channel Catfish varied from 234 to 676 mm in our sample (Table 21; Figure 28). Length distribution was represented by a PSD of 66 (Table 21). Average W_r of Channel Catfish was 99.

Collectively, salmonids caught in our sampling effort represented a broad range of growth potential and contributions to the fishery (Table 21; Figure 28). Brook Trout varied in length from 189 to 347 mm and their size structure was poor (PSD = 23). Comparatively, Brown Trout were characterized by larger size structure (PSD = 92) and varied in length from 240 to 573 mm. Rainbow Trout caught in our sample were primarily representative of recently stocked catchable length fish. Lengths of Rainbow Trout varied from 170 to 487 mm. Fin condition of these Rainbow Trout was generally noted to be deteriorated, reflecting the recent hatchery origin of these fish. A single Rainbow Trout and a single Rainbow Trout x Westslope Cutthroat Trout hybrid, not believed to be of hatchery origin and or representing fingerling stocking from 2017, were also caught in our sampling effort. No Westslope Cutthroat Trout of stock length or larger were caught in our sampling effort. Total length of Westslope Cutthroat Trout varied from 126 to 202 mm. Condition of all salmonid species, where estimated, was high with W_r varying from 96.6 to 100.4. Relative weight was not estimated for hatchery Rainbow Trout as condition was not considered to be reflective of in-lake influences. Similarly, relative weight was not estimated for wild-origin Rainbow Trout and Rainbow Trout x Westslope Cutthroat Trout hybrids because of low sample size.

Largemouth Bass exhibited rapid growth and good condition. Total length of collected fish varied from 178 to 476 mm and size structure was moderate (PSD = 68; Table 21; Figure 28). We estimated Largemouth Bass reached 305 mm by 4.2 years of age (Figure 30). Annual mortality of Largemouth Bass from age-4 to age-12 was low (17%; Figure 29), while condition was average (W_r = 100).

Although Smallmouth Bass were well-represented in our survey, size structure of the population was poor (PSD = 26). Total lengths varied from 76 to 453 mm and condition was average (W_r = 100; Table 21; Figure 28).

Length distribution of Black Crappie caught in our survey suggested few age classes were represented (Figure 28). Measured lengths varied from 174 mm to 344 mm and size structure was skewed toward larger individuals (PSD = 81). Black Crappie exhibited average condition (W_r = 97).

Comparing results of our lowland lake survey with prior surveys revealed several species-specific patterns (Table 23). Species composition and relative abundance (CPUE) were similar across surveys for multiple species. Others, such as Black Crappie and Channel Catfish, demonstrated increased representation. In contrast, Largemouth Bass and Yellow Perch were proportionally less common, despite relative abundance (CPUE, electrofishing) not demonstrating strong negative declines. Size structure and condition were not consistently described across surveys, making broad comparisons difficult. However, those with comparable values demonstrated both positive and negative changes. Channel Catfish and Largemouth Bass demonstrated dramatic increases in the proportion of larger individuals in the populations, while Yellow Perch demonstrated a decline. Relative weights generally remained at or above average values. However, we found Largemouth Bass condition improved with a near doubling of W_r.

Angler Survey

Anglers fished an estimated 19,733 hours on Cocolalla Lake between April 1, 2018 and March 31, 2019. Fishing occurred from boats and the shore. However, a majority (75%) of fishing effort was attributed to boat anglers. Angler effort peaked in June (3,905 hours; Figure 32). Minimal ice cover in December and January limited boat access and did not provide safe access for ice anglers. As a result, angler effort was lowest in December and January. Angler effort declined from June through September, but demonstrated a resurgence in October at 2,395 hours.

Catch rates varied widely by species and season (Table 24; Table 25). Anglers reported catching 11 species throughout the survey period. Angler catch rate was the highest for bass (0.75 fish/hour), and bass catch rates were highest in August. Individually, catch rates for Largemouth and Smallmouth Bass were similar at 0.36 and 0.39 fish/hour, respectively. Total catch rate for salmonids was 0.26 fish/hour, but varied widely by species. Rainbow Trout were caught at a rate of 0.14 fish/hour, while Brook Trout, Brown Trout, and Westslope Cutthroat Trout exhibited catch rates of 0.05 fish/hour or less. Rainbow Trout catch rate was highest in June and September. Peak catch rate for Brook Trout occurred in February, while Brown Trout fishing was most productive in April. Catch rate for Westslope Cutthroat Trout was comparably low throughout the year, with anglers experiencing the highest catch rate in October. Anglers experienced high catch rates for Yellow Perch relative to other species. Catch rates for Yellow Perch were high in June (1.01 fish/hour), February (0.36 fish/hour), and March (0.94 fish/hour). Channel Catfish were caught most readily in June at an estimated 0.24 fish/hour.

A majority of anglers fishing Cocolalla Lake were generalists and did not indicate they were specifically targeting any one species (Table 26). Of those anglers who did specify a targeted species, salmonids were collectively the most sought after. Bass also were targeted by a large proportion of anglers.

Catch from the Cocolalla Lake fishery largely reflected targeted effort. Smallmouth Bass, Largemouth Bass, Yellow Perch, and Rainbow Trout dominated the catch (Table 27). Channel Catfish and Pumpkinseed were both moderately represented in the catch. Harvest of fish caught varied by species, but was estimated to be less than 50% of fish caught for most species. More than 80% of bass, Pumpkinseed, Brook Trout, and Bluegill were released. Black Crappie, Channel Catfish, and Westslope Cutthroat Trout were the only species for which harvest was estimated to be greater than 50% of the total caught. All Westslope Cutthroat Trout reported were harvested.

DISCUSSION

Our lowland lake survey highlighted the diverse fish community in Cocolalla Lake. Collectively, the number of species and quality of fish represented is somewhat unique for mid-sized waters in the region. Although anglers did not target all species with regularity, the collective fishery provided diversity of opportunity for anglers.

The influence of hatchery products on the Cocolalla Lake fishery was evident in our survey. Hatchery-origin Channel Catfish, Rainbow Trout, and Westslope Cutthroat Trout collectively made up a large component of the catch. The contribution of hatchery products described in our survey was different than recent hatchery product evaluations and in part reflected changes in the use of hatchery Rainbow Trout. Relative abundance of hatchery Rainbow Trout stocked at fingerling length in spring was previously found to be low (Ryan et al. 2020c). Rainbow Trout, abundant in our survey, were thought to represent fish stocked prior to our survey in 2018 at a catchable length. We found Rainbow Trout stocked in 2018 not only were abundant in our survey, but persisted through the summer months and provided moderate catch rates post-stocking and into the fall and winter months. Based on the representation of hatchery Rainbow Trout from this survey, we recommend continued use of catchable-length Rainbow Trout. In contrast, we found Westslope Cutthroat Trout were not well-represented in our survey and those fish that were detected were small. Prior stocking evaluations suggested Westslope Cutthroat Trout stocked at fingerling length in Cocolalla Lake were moderately abundant and achieved relatively large length (> 400 mm; Ryan et al. 2020c). Westslope Cutthroat Trout stocking densities were variable in the years prior to this survey (IDFG unpublished data) making it difficult to parse out how stocking rate versus survival may have influenced the abundance of Westslope Cutthroat Trout at the time of our survey. As such, we recommend periodic monitoring of Westslope Cutthroat Trout relative abundance to better understand if changes in return to creel have occurred or if other factors (e.g., survey timing) influenced our interpretation of abundance in the lake.

Relative abundance of Channel Catfish in Cocolalla Lake was greater than previously reported and may reflect recent changes in stocking densities. We also described a notable increase in PSD, suggesting the proportion of Channel Catfish quality length (410 mm) and larger increased (Fredericks et al. 2009, Fredericks et al. 2013). Channel Catfish stocking density was reduced from approximately 24 to 12 fish/hectare beginning in 2014 (IDFG, unpublished data). Stocking density and lake productivity have been shown to influence survival, growth, and condition of Channel Catfish in small impoundments (Michaletz 2009). As such, shifts in relative abundance and size structure depicted in our survey may reflect changes in stocking density. We did not evaluate survival or growth and were not able to evaluate how changes in dynamic rates may have influenced abundance and/or size structure. We recommend future sampling efforts incorporate these metrics to better understand the dynamics of this population as they relate to stocking.

Prior investigators suggested standard sampling methods were not adequate to effectively describe Channel Catfish size structure or population dynamics (Fredericks et al. 2013, Carter-Lynn et al. 2015). Although we were unable to critically evaluate how size structure in our sample reflected that in the population, we did collect a robust sample that provided confidence in generally describing the population. We found sinking gill nets were the most efficient method of capture in our survey, but all sampling methods caught Channel Catfish. Our survey differed from the previous lowland lake survey effort by incorporating additional sampling effort (Fredericks et al. 2009).

The relative abundance of Channel Catfish described in our population survey was not reflected in the fishery. Channel Catfish were the second most abundant species detected among all sampling techniques, but angler catch rates and corresponding total catch was low to moderate relative to other abundant species (i.e., Yellow Perch). Although Channel Catfish were abundant, we interviewed few anglers who targeted them. This suggests they may be underutilized. Alternatively, our angler survey design may not have effectively described Channel Catfish angling effort or success. Traditional angler surveys conducted during daylight hours may not detect Channel Catfish anglers that primarily fish at night (Davis et al. 1996, Fredericks et al. 2013). Irrespective of our angler survey, our findings were consistent with prior estimates of exploitation. Fredericks et al. (2013) were unable to detect exploitation of Cocolalla Lake Channel Catfish at any level. Given Channel Catfish were abundant in Cocolalla Lake, but few potential anglers targeted them, we recommend stocking density be considered as a management tool to balance the quality of the fishing opportunity with angler interest level. In addition, a public outreach campaign may be beneficial to inform anglers this type of fishery exists in the region.

Smallmouth Bass represented a new species in Cocolalla Lake. The origin of these fish is unknown. We found no record of intentional introduction and no illegal introduction event was known to have occurred. Cocolalla Creek does provided a potential migratory corridor to the Pend Oreille River where Smallmouth Bass have been present since the late 1990s, potentially providing a source population (See Pend Oreille River chapter in this report). Our angler survey suggested Smallmouth Bass provided a fishery component similar to Largemouth Bass and shared similar catch rates. However, size structure of the population was less robust as evident by reported PSD. Although, Smallmouth Bass provide a new fishery opportunity, they may also negatively influence species composition in the future. Notable changes in species presence and abundance has been anecdotally noted following the expansion of Smallmouth Bass in other waters (See Pend Oreille River chapter in this report, Maiolie et al. 2011).

Size structure of Largemouth Bass in Cocolalla Lake increased dramatically from the most recent historical survey. Comparisons of electrofishing catch rate suggested abundance was similar to prior survey efforts and did not contribute to shifts in size structure. Although infrequent angler catch and use data was available for comparison, we found no evidence that angler harvest previously influenced size structure. Davis et al. (1996) found no anglers reported catch or harvest of Largemouth Bass in their 1992 survey. Comparably, we found harvest of Largemouth Bass was low and under 10% of total catch. No description of survival and growth was available from prior surveys to evaluate how changes in dynamic rates influenced the population. We recommend future sampling continue to incorporate these metrics to better understand the dynamics of this population.

Yellow Perch were less abundant than described in prior surveys. Age structure of the population suggested a pattern of pulsed recruitment may be occurring, influencing abundance and age class representation. Cyclic dynamics, where individual age classes dominate a Yellow Perch population for multiple years, have been observed in other populations (Sanderson et al. 1999). Similar patterns of recruitment were noted in the Pend Oreille system (Watkins et al. 2018). This observation suggests fishing for larger Yellow Perch may be inconsistent from year-to-year.

Anecdotal angler observations generally confirm inconsistent Yellow Perch fisheries are typical in northern Idaho waters.

Bluegill were not detected in our fish community survey despite being identified in a previous fish community survey (Fredericks et al. 2009). It is unclear whether a short-term introduction occurred or if prior observations represented a misidentification. Bluegill were also reported by anglers interviewed during our angler survey. However, no Bluegill were harvested and observed by creel clerks. Pumpkinseed were relatively abundant in our fish community survey and were previously observed in other surveys (Davis et al. 1996, Fredericks et al. 2009). We hypothesize reported catches of Bluegill likely were misidentified Pumpkinseed.

Angler use of the Cocolalla Lake fishery more than doubled since the early 1990s. An estimated 8,877 hours of angler effort were occurred in 1992 from April to September (Davis et al. 1996). Had ice fishing conditions developed in 2018-19, the increase in effort would have been even more substantial. We speculate that more diverse angler interests may have contributed to increased effort in the fishery. For example, we found anglers continued to pursue salmonids as their primary target in the fishery, but also targeted bass with regularity. No catch of Largemouth Bass was reported in 1992 despite being abundant at that time (Davis et al. 1996). Catch of bass (Largemouth and Smallmouth bass) was the highest of all species detected in our survey.

Angler effort on Cocolalla Lake increased from previously estimated levels, but remained low relative to other area lakes. For example, angler effort on Spirit Lake was 45,235 hours over the same time period as our survey (see “Spirit Lake Creel Survey” chapter of this report). Annual angler effort on Hayden Lake in 2010 was estimated at 74,000 hours (Maiolie et al 2011). A number of factors potentially influenced differences in angler effort between waters such as waterbody size, location, species composition, fish abundance, population size structure, and accessibility. Although all of these factors may have been influential on Cocolalla Lake, accessibility likely influence angler effort more than most. Boat access for medium to large boats in the summer and fall months was challenging at the single public boat ramp due to very shallow water surrounding the site and limiting use. Winter fishery opportunities may also be influenced by accessibility due to ice conditions. Safe ice cover, allowing for ice angling opportunity, was only available for a short period during the 2018/19 period potentially reducing overall effort. Although angler effort in Cocolalla Lake may be influenced by a number of potential factors, effort levels in prior surveys demonstrated proportionally similar deviation from other lakes in the area suggesting influencing factors may be common from year-to-year (Davis et al. 1996).

The fish community of Cocolalla Lake as described in our survey was diverse and we found the resulting fishery in 2018 reflected this diversity. Although the fish community described in the early 1990s was similar relative to species composition, the fishery at that time was primarily salmonid focused and did not reflect the existing fish community (Davis et al 1996). For example, no catch of Largemouth Bass or Black Crappie was reported by anglers in 1992 despite being comparably abundant at that time. We also found catch rates of Channel Catfish (0.05 fish/hour) and Yellow Perch (0.07 fish/hour) in 1992 were lower than described in our survey. In contrast, catch rates of Rainbow Trout were similar both surveys. The period included in the 1992 angler survey was different (i.e., April – September; Davis et al. 1996) and may have influenced our comparison of catch rates. Regardless, this comparison suggests angler preferences have changed over time and/or new anglers with differing interest are utilizing the fishery. As such, we recommend management of the Cocolalla Lake fishery focus on maintaining a diverse fish community to support current angler interests.

Our angler survey design had limitations that may have influenced our results. For example, we only interviewed anglers at the IDFG access site. As such, anglers accessing the lake from alternative locations, such as private residences, were not included in our survey. While we made an assumption, angler catch rates did not vary by where anglers accessed the fishery, we did not test our assumption. We also experienced difficulty completing roving counts of anglers during the winter ice fishery and relied on available viewpoints to make counts. In addition, ice anglers commonly used shelters while fishing making it difficult to count individual anglers. While we made an attempt to confirm angler counts during completed trip interviews, not all anglers completed their trip during a survey period. This limitation had the potential to underestimate angler effort during the winter ice fishery period.

RECOMMENDATIONS

1. Continue the use of catchable length Rainbow Trout to enhance the salmonid fishery in Cocolalla Lake.

2. Periodically monitor relative abundance of Westslope Cutthroat Trout to better understand how they contribute to the Cocolalla Lake fishery, how stocking density impacts abundance, and how factors such as survey timing influence our interpretation of their abundance in the lake.

3. More broadly incorporate measures of survival and growth in future investigations of the Cocolalla Lake fish community to better understand how population dynamics relate to species interactions and hatchery supplementation.

4. Re-evaluate Channel Catfish stocking density by considering the balance between quality of fishing opportunity in Cocolalla Lake and angler interest level.

5. Promote the Channel Catfish fishery in Cocolalla Lake to make anglers aware of this unique regional fishery opportunity.

6. Focus management of the Cocolalla Lake fishery on maintaining a diverse fish community to support current angler interests.

Table 20. Sample locations by date and method from a 2018 lowland lake survey of Cocolalla Lake, Idaho.

Unit	Date	Method	Time (h:mm)	Latitude	Longitude	Datum
E1	5/31/2018	Boat Electrofishing	0:11	48.117815	-116.614900	WGS84
E2	5/31/2018	Boat Electrofishing	0:10	48.111174	-116.618950	WGS84
E3	5/31/2018	Boat Electrofishing	0:10	48.130317	-116.621170	WGS84
E4	5/31/2018	Boat Electrofishing	0:15	48.113479	-116.624792	WGS84
E5	5/31/2018	Boat Electrofishing	0:10	48.109976	-116.626393	WGS84
E6	5/31/2018	Boat Electrofishing	0:10	48.108730	-116.624997	WGS84
E7	5/31/2018	Boat Electrofishing	0:10	48.122401	-116.626294	WGS84
E8	5/31/2018	Boat Electrofishing	0:11	48.138267	-116.616096	WGS84
E9	5/31/2018	Boat Electrofishing	0:10	48.134791	-116.601638	WGS84
F1	5/29/2018	Floating Gill Net	23:10	48.131122	-116.610956	WGS84
F2	5/29/2018	Floating Gill Net	21:00	48.128678	-116.608085	WGS84
F3	5/30/2018	Floating Gill Net	18:30	48.116702	-116.619055	WGS84
F4	5/29/2018	Floating Gill Net	21:35	48.133565	-116.617005	WGS84
F5	5/30/2018	Floating Gill Net	18:55	48.133858	-116.611006	WGS84
S1	5/30/2018	Sinking Gill Net	15:50	48.125098	-116.625358	WGS84
S2	5/29/2018	Sinking Gill Net	29:25	48.122105	-116.616286	WGS84
S3	5/29/2018	Sinking Gill Net	15:50	48.123791	-116.613518	WGS84
S4	5/30/2018	Sinking Gill Net	16:40	48.127439	-116.621668	WGS84
S5	5/29/2018	Sinking Gill Net	19:20	48.118629	-116.616388	WGS84
T1	6/4/2018	Trap Net	24:15	48.130170	-116.606543	WGS84
T2	6/5/2018	Trap Net	22:40	48.114133	-116.616642	WGS84
T3	6/5/2018	Trap Net	23:10	48.108936	-116.625712	WGS84
T4	6/5/2018	Trap Net	23:15	48.129895	-116.621462	WGS84
T5	6/5/2018	Trap Net	23:08	48.116783	-116.624689	WGS84
T6	6/4/2018	Trap Net	24:35	48.115131	-116.615925	WGS84
T7	6/4/2018	Trap Net	24:25	48.126763	-116.609876	WGS84
T8	6/4/2018	Trap Net	23:55	48.137431	-116.601620	WGS84
T9	6/5/2018	Trap Net	23:20	48.134713	-116.617155	WGS84
T10	6/4/2018	Trap Net	24:25	48.128278	-116.608184	WGS84

Table 21. Descriptive statistics for samples collected from Cocolalla Lake, Idaho during a lowland lake survey in May and June 2018. Statistics summarized include catch, proportion of catch by number and biomass, minimum and maximum total length (TL, mm), proportional stock density (PSD), and relative weight (W_r; SD).

Species	Catch	% Count	% Biomass	Mean TL	Max TL	PSD	W_r
Black Crappie	99	8%	5%	236	344	80.8	100 (1.9)
Brook Trout	31	3%	2%	266	347	23.3	99 (2.0)
Brown Bullhead	47	4%	3%	273	312	97.9	99 (1.1)
Brown Trout	12	1%	3%	467	573	91.7	97 (1.8)
Channel Catfish	243	20%	41%	426	676	66.4	99 (2.4)
Hatchery Rainbow Trout	110	9%	3%	226	487	n/a	--
Largemouth Bass	52	4%	7%	328	476	68.0	100 (1.6)
Largescale Sucker	116	9%	23%	438	561	n/a	97 (1.5)
Longnose Sucker	30	2%	4%	341	576	n/a	97 (2.0
Peamouth	15	1%	1%	291	315	n/a	--
Pumpkinseed	60	5%	1%	130	201	18.3	105 (3.6)
Rainbow Trout	1	< 1%	< 1%	193	193	--	--
Rainbow Trout x Cutthroat Trout Hybrid	1	< 1%	< 1%	223	223	--	--
Smallmouth Bass	85	7%	5%	245	453	26.1	100 (1.5)
Westslope Cutthroat Trout	9	1%	< 1%	161	202	0.0	100 (2.1)
Yellow Perch	320	26%	4%	157	253	14.8	97 (4.0)

Table 22. Catch per unit effort (1 SD) from electrofishing (fish/hour), floating gill net (fish/net), sinking gill net (fish/net), and trap net (fish/net) effort on Cocolalla Lake, Idaho in May and June 2018.

Species	Electrofishing	Floating Gill Net	Sinking Gill Net	Trap Net
Black Crappie	1.3 (4)	0.2 (0.4)	17.4 (8.0)	1.0 (2.7)
Brook Trout	0.0	2.4 (4.3)	3.6 (2.3)	0.1 (.3)
Brown Bullhead	8.7 (14.1)	0.0	0.8 (1.3)	3.3 (4.7)
Brown Trout	0.0	0.6 (0.9)	1.8 (1.9)	0.0
Channel Catfish	10.0 (11.2)	14 (8.2)	27.4 (5.7)	2.2 (3.8)
Hatchery Rainbow Trout	58.6 (76.6)	0.2 (0.4)	4.2 (1.3)	0.0
Largemouth Bass	28.6 (27)	0.2 (0.4)	1.6 (1.5)	0.0
Longnose Sucker	10.7 (25.6)	0.0	1.0 (1.7)	1.0 (3.0)
Largescale Sucker	33.3 (29.7)	2.0 (4.5)	4.8 (4.3)	3.1 (4.1)
Peamouth	0.0	2.6 (2.4)	0.4 (0.9)	0.0
Pumpkinseed	12.0 (17.4)	0.0	0.4 (0.5)	4.4 (7.1)
Rainbow Trout	0.7 (2)	0.0	0.0	0.0
Rainbow Trout x Cutthroat Trout Hybrid	0.0	0.2 (0.4)	0.0	0.0
Smallmouth Bass	16.0 (17.7)	0.0	11.8 (8.2)	0.2 (0.7)
Westslope Cutthroat Trout	2.7 (3.2)	0.2 (0.4)	0.6 (1.3)	0.1 (0.3)
Yellow Perch	127.9 (77.4)	0.6 (1.3)	5.2 (5.4)	11 (16.3)

Table 23. Summary of species-specific metrics from past and present surveys of Cocolalla Lake. Metrics include percent of total catch from all gear types by number (% Catch), percent of total catch from all gear types by weight (% Biomass), electrofishing fish/hour (CPUE), proportional stock density (PSD), and mean relative weight (W_r).

Year	Species	% of Catch	% of Biomass	CPUE	PSD	W_r
1992	Black Crappie	4%	--	5	--	--
2008	Black Crappie	≤1%	≤1%	11	--	--
2018	Black Crappie	8%	5%	1	81	100
1992	Brook Trout	--	--	--	--	--
2008	Brook Trout	≤1%	2%	8	--	--
2018	Brook Trout	3%	2%	--	23	99
1992	Brown Trout	1%	--	3	--	--
2008	Brown Trout	≤1%	5%	9	--	--
2018	Brown Trout	1%	3%	--	92	97
1992	Brown Bullhead	1%	--	--	--	--
2008	Brown Bullhead	3%	4%	31	--	--
2018	Brown Bullhead	4%	3%	9	98	99
1992	Channel Catfish	11%	--	9	--	--
2008	Channel Catfish	5%	12%	--	5	105
2018	Channel Catfish	20%	41%	14	66	99
1992	Largemouth Bass	5%	--	21.0	--	--
2008	Largemouth Bass	11%	19%	33.5	6	54
2018	Largemouth Bass	4%	7%	29.0	68	100
1992	Pumpkinseed	6%	--	8	--	--
2008	Pumpkinseed	11%	≤1%	171	--	--
2018	Pumpkinseed	5%	1%	12	18	105
1992	Rainbow Trout	8%	--	35	--	--
2008	Rainbow Trout	≤1%	2%	4	--	--
2018	Rainbow Trout	10%	4%	1	--	--
1992	Smallmouth Bass	--	--	--	--	--
2008	Smallmouth Bass	--	--	--	--	--
2018	Smallmouth Bass	7%	5%	16	26	100
1992	Yellow Perch	57%	--	386	--	--
2008	Yellow Perch	51%	8%	153	33	85
2018	Yellow Perch	26%	4%	128	15	97
1992	Westslope Cutthroat Trout	--	--	--	--	--

Table 23 (continued)

Year	Species	% of Catch	% of Biomass	CPUE	PSD	W_r
2008	Westslope Cutthroat Trout	≤1%	2%	5	--	--
2018	Westslope Cutthroat Trout	1%	≤0.01	3	--	100
1992	Peamouth	2%	--	--	--	--
2008	Peamouth	3%	≤1%	1	--	--
2018	Peamouth	1%	1%	--	--	--
1992	Largescale Sucker	6%	--	--	--	--
2008	Largescale Sucker	10%	37%	96	--	--
2018	Largescale Sucker	9%	23%	33	--	97
1992	Longnose Sucker	--	--	--	--	--
2008	Longnose Sucker	3%	9%	28	--	--
2018	Longnose Sucker	2%	4%	11	--	97

Table 24. Survey-wide catch rate by species estimated from angler interview data collected between April 1, 2018 and March 31, 2019 on Cocolalla Lake, Idaho.

Species	Total Catch Reported	Catch Rate
Black Crappie	13	0.04
Bluegill	1	<0.01
Brook Trout	12	0.04
Brown Trout	17	0.05
Channel Catfish	34	0.11
Largemouth Bass	113	0.36
Pumpkinseed	17	0.05
Rainbow Trout	44	0.14
Smallmouth Bass	123	0.39
Westslope Cutthroat Trout	8	0.03
Yellow Perch	76	0.24
Trout (Combined)	81	0.26
Bass (Combined)	236	0.75

Table 25. Catch rate by species and month for fish reported by anglers from April 1, 2018 through March 31, 2019 on Cocolalla Lake, Idaho.

Species	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar
Black Crappie	--	0.15	--	0.04	0.04	--	--	--	--	--	--	--
Bluegill	--	--	--	--	0.02	--	--	--	--	--	--	--
Brook Trout	0.09	--	--	--	--	--	--	0.25	--	--	0.36	--
Brown Trout	0.28	0.15	0.02	--	--	--	0.03	--	--	--		--
Channel Catfish	--	0.10	0.24	0.07	0.15	--	0.14	--	--	--	0.04	--
Largemouth Bass	--	0.09	0.51	0.21	1.27	0.20	0.07	--	--	--	--	--
Pumpkinseed	--	--	0.36	--	--	--	--	--	--	--	--	--
Rainbow Trout	0.09	0.14	0.24	0.14	0.06	0.29	0.14	--	--	--	0.20	--
Smallmouth Bass	0.14	0.31	0.41	0.25	1.19	0.10	0.14	--	--	--	--	--
Westslope Cutthroat Trout	0.09	0.02	0.02	--	--	--	0.14	--	--	--	--	--
Yellow Perch	--	--	1.01	--	0.13	0.29	0.07	--	--	--	0.36	0.94

Table 26. Proportion of angler reported targets by species or species group from anglers interviewed at Cocolalla Lake, Idaho from April 1, 2018 through March 31, 2019.

Species	% Anglers Targeting
Bass	15%
Black Crappie	2%
Bullhead	1%
Channel Catfish	2%
Trout	31%
Yellow Perch	4%
General	45%

Table 27. Catch and harvested proportion of catch by species from Cocolalla Lake, Idaho from April 1, 2018 through March 31, 2019.

Species	Total Catch	% Harvest
Black Crappie	520	56.1%
Bluegill	100	0.0%
Brook Trout	579	0.0%
Brown Trout	793	41.4%
Channel Catfish	1,209	63.8%
Largemouth Bass	5,677	9.5%
Pumpkinseed	955	11.8%
Rainbow Trout	3,867	32.3%
Smallmouth Bass	5,807	11.6%
Westslope Cutthroat Trout	582	100.0%
Yellow Perch	4,901	38.9%

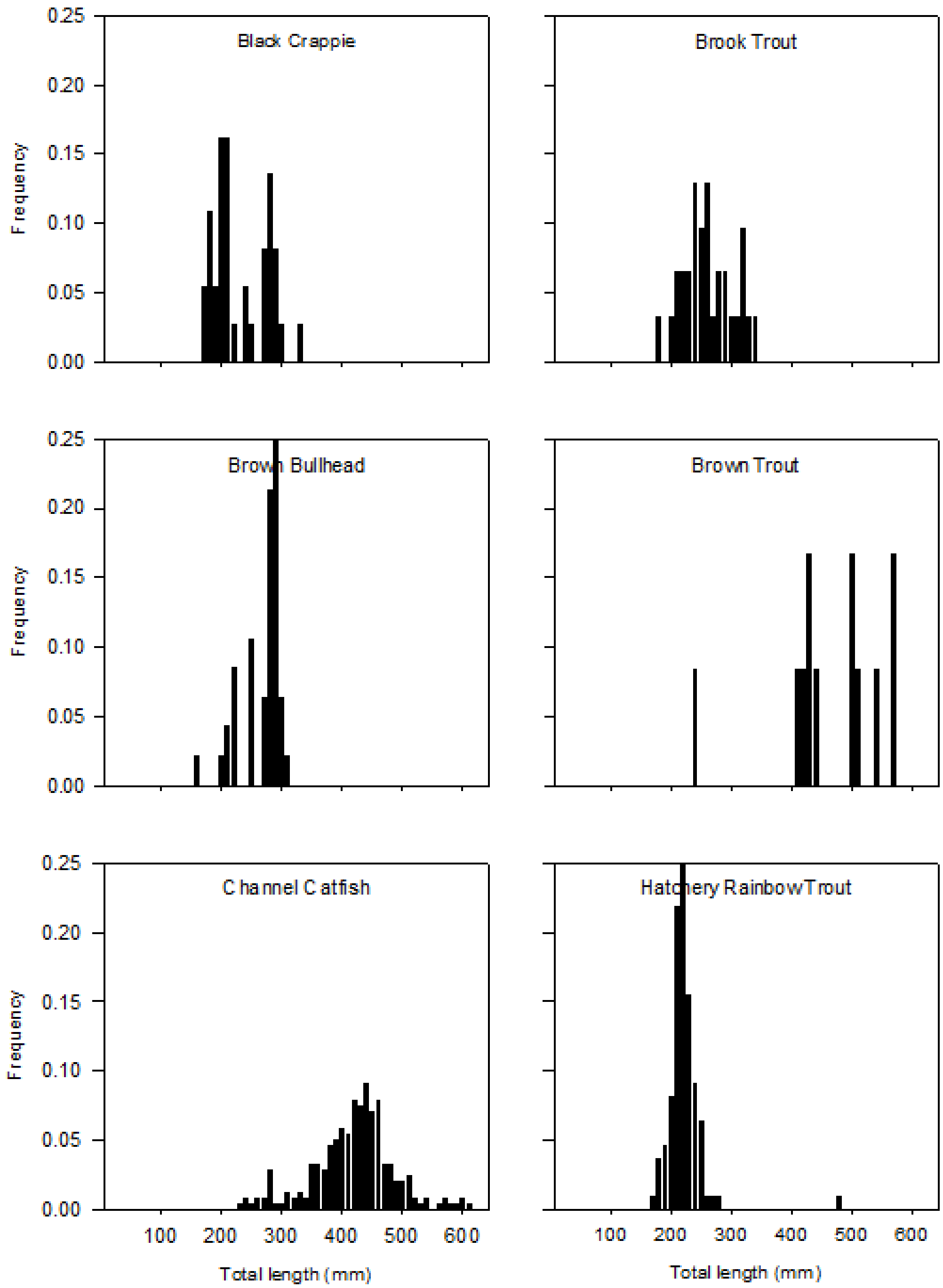
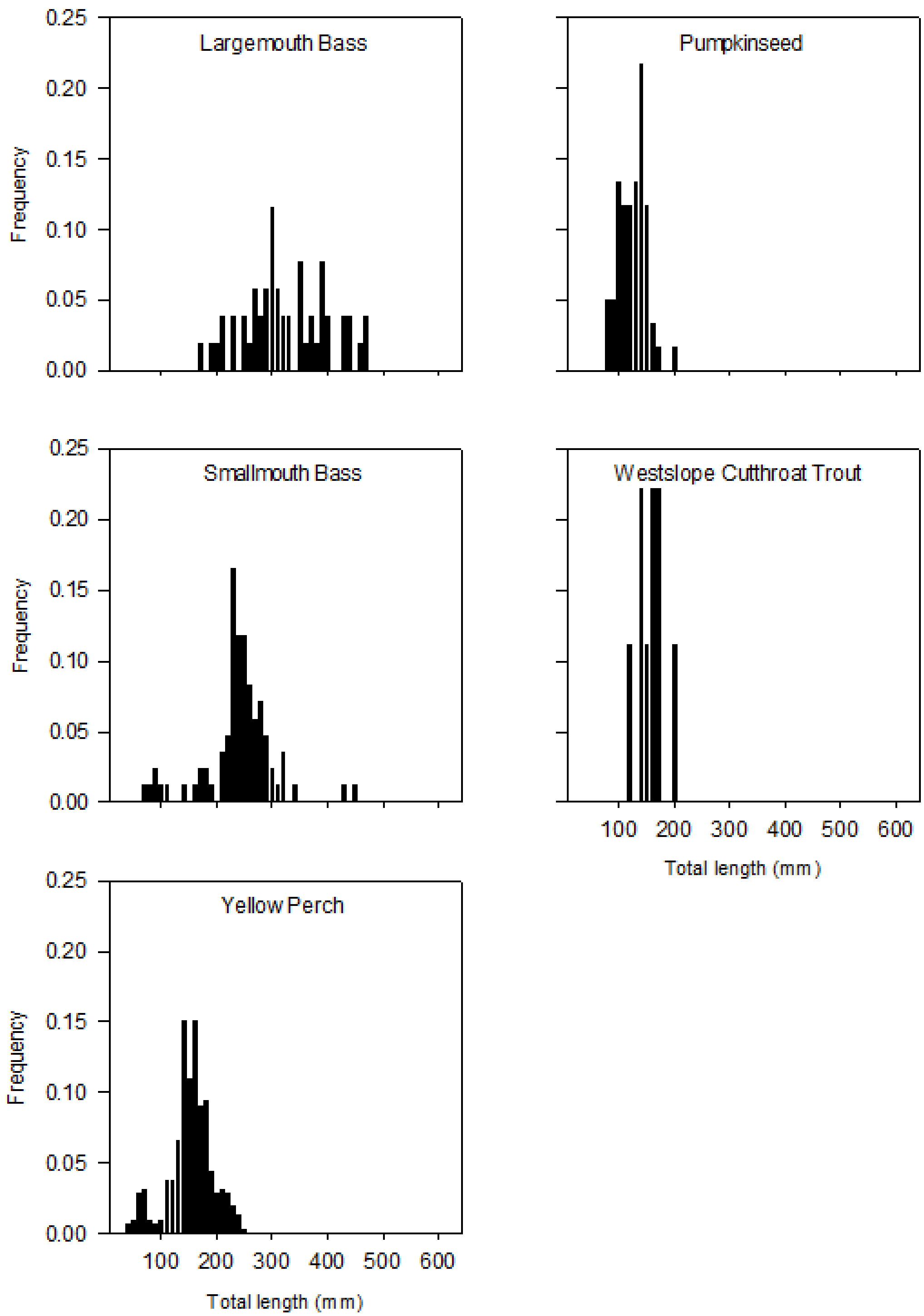


Figure 28. Relative length-frequency distributions (%) of game fish sampled using boat electrofishing, gill nets, and trap nets from Cocolalla Lake, Idaho in June 2018.

Figure 28 (continued)



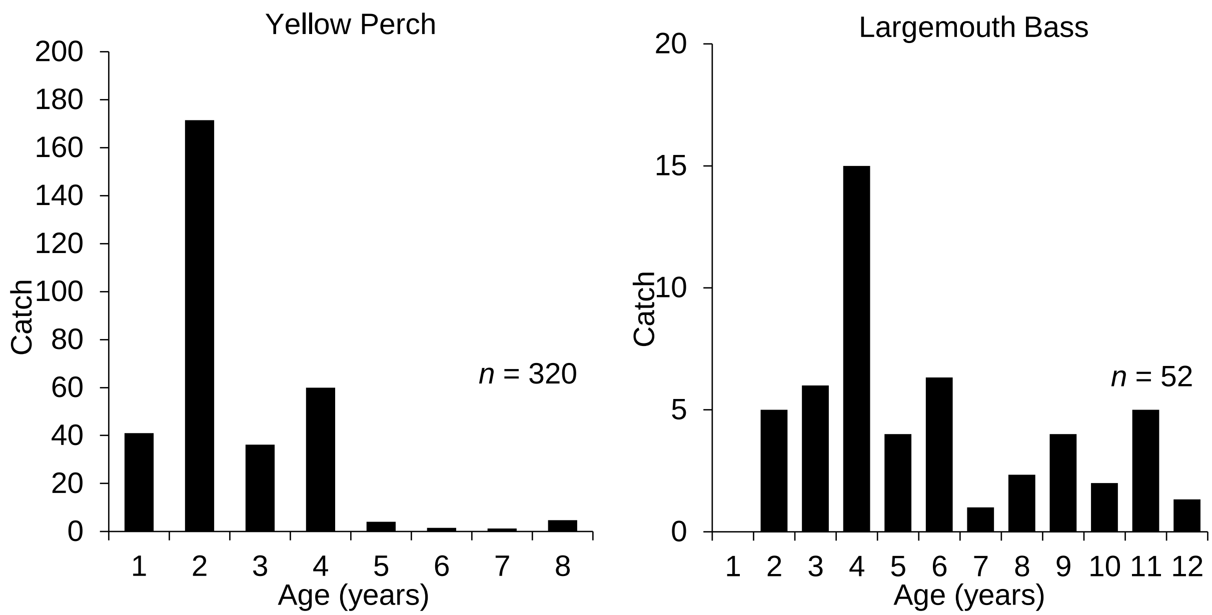


Figure 29. Age frequency of Yellow Perch and Largemouth Bass sampled from Cocolalla Lake in May and June 2018.

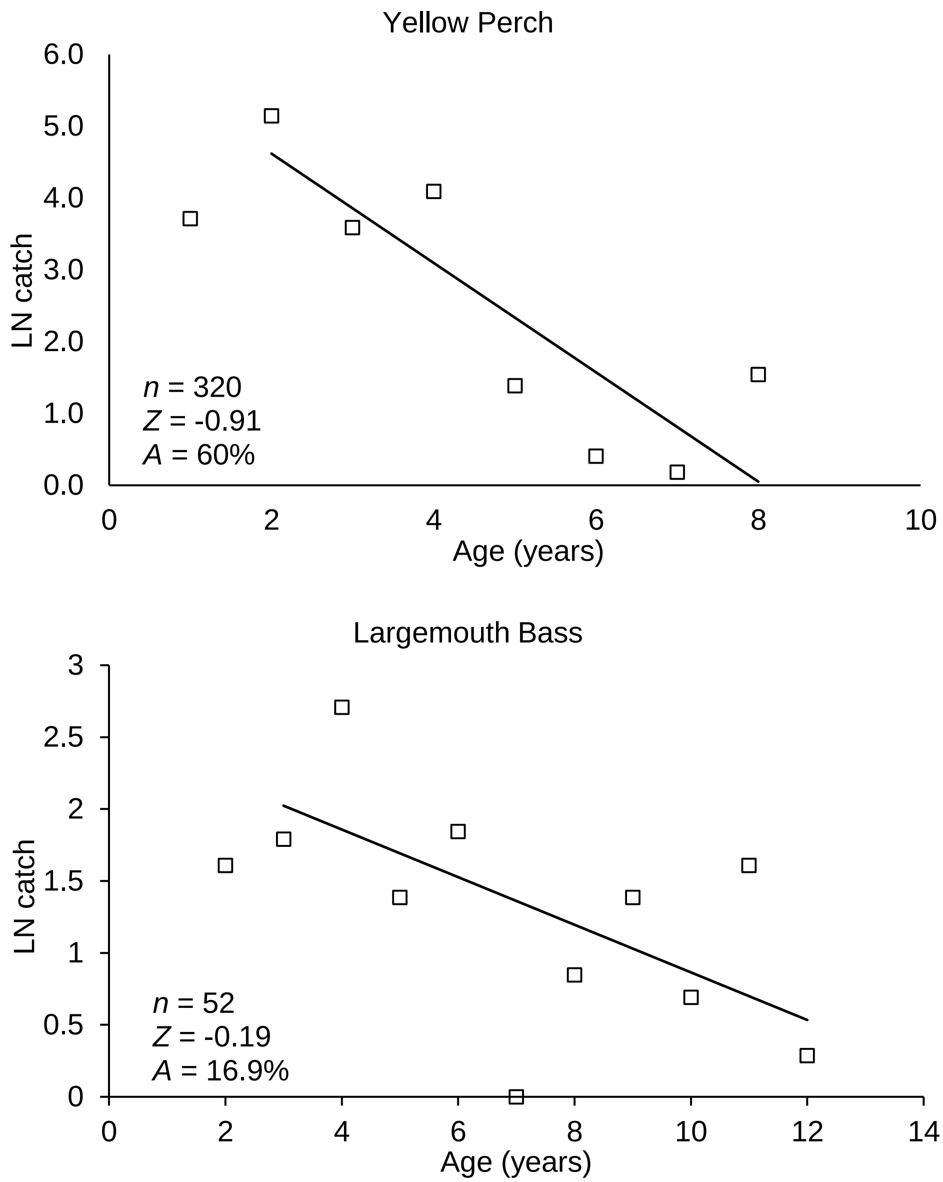


Figure 30. Catch curves developed from catch-at-age data for Yellow Perch and Largemouth Bass sampled in Cocolalla Lake in May and June 2018.

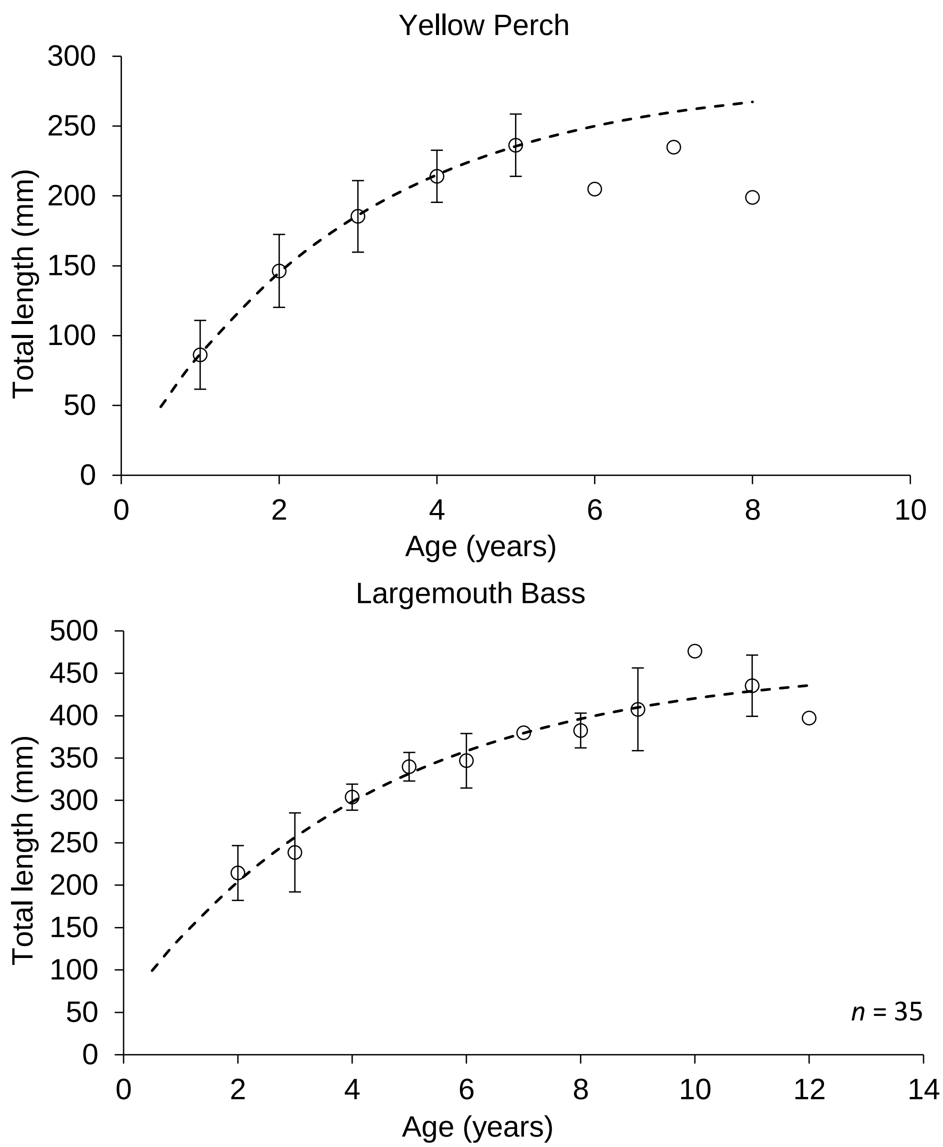


Figure 31. Mean length-at-age at time of capture (± 1 SD) estimated from Yellow Perch and Largemouth Bass sampled in Cocolalla Lake in May and June 2018.

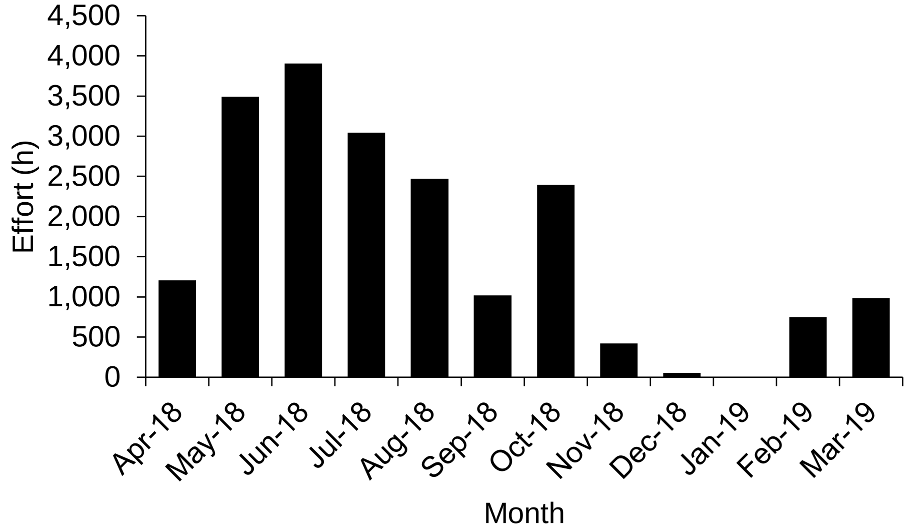


Figure 32. Angler effort by month estimated from Cocolalla Lake, Idaho from April 2018 through March 2019.

HAYDEN LAKE INVESTIGATIONS

Hayden Lake, located northeast of Hayden, Idaho in the Panhandle Region, provides fishing opportunity for multiple fish species and is a popular fishing destination. Rainbow Trout *Oncorhynchus mykiss* have been stocked in Hayden Lake since the early 1900s and have historically provided a quality fishery, but represent only a small portion of the effort and catch in recent years. Identifying the cause and remedy for declining trout fishing opportunities in Hayden Lake has been an ongoing focus of fisheries managers. Kokanee have also been stocked in Hayden Lake at low density (62-93 fish/ha) since 2011 to provide a pelagic fishery. However, production from wild-spawning kokanee may influence abundance and subsequent growth rates, making it difficult to maintain a quality kokanee fishery. In 2018, we continued to evaluate Rainbow Trout stocking in an effort to identify stocking strategies that improve the fishery. We used standardized floating experimental gill nets to describe relative abundance of Rainbow Trout in the lake post-stocking. We also monitored kokanee origin to describe the level of wild production occurring. Suspended gill nets were used to collect kokanee. Kokanee otoliths were examined for thermal marks to identify hatchery and wild origin fish. We collected a single Rainbow Trout in our sampling, suggesting abundance of stocked fish was low. Unmarked kokanee made up a small proportion of our sample, suggesting wild origin fish were rare. We recommend the evaluation of fingerling Rainbow Trout stocking be continued. We also recommend periodic monitoring of wild kokanee production in Hayden Lake continue. Because wild origin kokanee were not abundant, we did not recommend efforts be made to limit kokanee spawning in Hayden Lake tributaries.

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INTRODUCTION

Hayden Lake, located northeast of Hayden, Idaho in the Panhandle Region, provides fishing opportunity for multiple fish species and is a popular destination for anglers. A mix of warmwater species, including Largemouth Bass *Micropterus salmoides*, Black Crappie *Pomoxis nigromaculatus*, and Yellow Perch *Perca flavescens* were introduced in the early 1900s and are the primary focus of anglers (Maiolie et al 2011). More recent sportfish introductions in Hayden Lake also provide popular fishing opportunities. Smallmouth Bass *Micropterus dolomieui*, legally introduced, and Northern Pike *Esox lucius*, illegally introduced, added to popular littoral fisheries (Maiolie et al. 2011). Kokanee *Oncorhynchus nerka* stocked since 2011 have noticeably increased angling effort in the pelagic areas of the lake. Historically, Hayden Lake provided a popular fishery for native Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi*, but abundance declined and they are now rare in angler catches (Mauser 1978, Maiolie et al. 2011). In more recent years, the discontinuation of Westslope Cutthroat Trout stocking further impacted cutthroat fishing opportunity. Rainbow Trout were stocked in Hayden Lake since the early 1900s and angler reports suggested stocking historically provided a quality fishery. However, Rainbow Trout catch rates were poor (≤ 0.3 fish/hr) throughout the history of formal angler surveys on the lake (Ellis 1983, Davis et al. 2000, Maiolie et al. 2011). Rainbow Trout represented only a small portion of the targeted effort and catch in recent years. The lack of emphasis on the Rainbow Trout fishery is likely due in part to the quality of the fishery.

Improvement of the Hayden Lake trout fishery has been an ongoing focus of fisheries managers. Multiple management actions have been attempted to increase trout survival and abundance. Management actions included introduction of mysid shrimp *Mysis diluviana* (mysids) an alternative food source (Heimer 1970), stocking rate manipulations, and experimentation with stocked strains and stocking locations. Despite these efforts, angler catch rates on trout continue to be low (Maiolie et al. 2011).

Kokanee have been stocked at low density (62-93 fish/ha) in Hayden Lake since 2011 to provide a pelagic sport fishery component (IDFG, unpublished data). Low density stocking was intended to provide a balance in size and abundance. Early-strain kokanee were stocked in most years (except 2018) and have performed well with average total length of age-2 fish in the spring varying from 289 to 388 mm (IDFG unpublished data). Although observed kokanee growth was desirable, some concern existed over maintenance of desired growth rates. This concern existed in part because mature kokanee have strayed to lake tributaries to spawn since introduction, but survival and associated production to the lake is not known. Production from wild-spawning kokanee may influence abundance and subsequent growth rates, making it difficult to maintain a quality kokanee fishery.

Rainbow Trout stocking evaluations have been an ongoing project. In 2018, we continued Rainbow Trout stocking evaluations and kokanee monitoring in an effort to understand and improve the Hayden Lake fishery. Recent investigations included evaluations of stocked Rainbow Trout origin (i.e., Cabinet Gorge Hatchery), strain, and density. We also monitored kokanee origin in the lake to describe what level of wild production has occurred.

OBJECTIVES

1. Estimate relative return of Rainbow Trout stocked as fall fingerlings
2. Describe wild kokanee production

METHODS

Rainbow Trout Stocking Evaluation

We described relative abundance of hatchery Rainbow Trout in Hayden Lake using catch rates in standard floating experimental gill nets (IDFG 2012). Twenty-four nets were fished overnight in Hayden Lake on May 2 and 3 as well as June 12 and 13, 2018. Netting effort was completed in June because May effort resulted in low catch and observed cold water temperatures (9-12 °C) thought to potentially influence catch rate. June water temperature was warmer at 17°C. Net locations were replications of random sites selected for a 2014 survey (Table 28). All nets were fished overnight. We reported mean catch per unit effort (CPUE, fish/net) as a measure of relative abundance. We identified all fish, measured total length (mm), and checked individuals for marks.

We intended to use proportional differences in relative abundance to explore the success of different Rainbow Trout stocking groups in Hayden Lake (Table 29). Marks were not available to distinguish every stocking group. As such, we anticipated also using fish lengths and fin condition to allow coarse identification of stocked groups. Unique thermal marks, applied during early hatchery rearing, were used to distinguish the 2017 cohort.

Kokanee Monitoring

We described the production of wild kokanee in Hayden Lake by estimating the proportion of wild origin kokanee. A sample of kokanee was collected using suspended gill nets as described by Klein et al. (2019). Gill nets were 48.8 m long and 6.0 m in depth with 16 panels that were each 3.0 m long. Each net was configured with eight mesh sizes, including 12.7-, 19.0-, 25.4-, 38.1-, 50.8-, 63.5-, 76.2-, and 101.6-mm stretch measure. Two sample locations were non-randomly selected based on prior knowledge of kokanee distribution in the lake (Table 30). Multiple nets were suspended at each location at varying depths to cover the vertical distribution of kokanee in the water column. All nets were fished overnight. Captured fish were identified, measured to total length (mm), and otoliths were removed.

Kokanee otoliths were inspected for thermal marks to identify hatchery and wild origin fish. Thermal marks were applied at the IDFG Cabinet Gorge Fish Hatchery during early hatchery rearing by manipulating water temperature in a designated pattern. Thermally marked patterns of growth were visible on otolith structures and appeared as banding patterns with the thickness and separation of bands influenced by the timing and duration of water temperature manipulation. Thermal mark patterns were unique to each year class. Identification of marks was completed by mounting otoliths, sulcus side up, to glass slides with Crystalbond 509 (Electron Microscopy Products, Hatfield, PA). Otoliths were then sanded until clearly viewable near the origin and viewed under 100x to 200x magnification to identify whether a mark was present. We assigned age to individual kokanee using thermal mark patterns. Age was assigned by length for those fish without a detectable thermal mark.

RESULTS

Rainbow Trout Stocking Evaluation

One Rainbow Trout was captured among all gillnetting efforts. Based on appearance (fin erosion), this fish was believed to be of hatchery origin. Total length of the Rainbow Trout caught suggested the fish was from the 2015 or 2016 outplant. Bycatch in our sampling included Black Crappie *Pomoxis nigromaculatus*, Brown Bullhead *Ictalurus nebulosus*, kokanee, Northern Pike, Tench *Tinca tinca*, and Westslope Cutthroat Trout (Table 31). Catch rate was highest for kokanee (1.1 ± 0.5 fish/net; 80% CI).

Kokanee Monitoring

We caught 25 kokanee among all gill net sets. Otoliths from 24 fish were examined for marks. Thermal marks were not detected on eight percent of the catch. Kokanee from age-0 to age-2 were represented in our sample. Age-1 fish represented 56% of the catch and age-2 fish represented 40% of the catch. Only one age-0 fish was collected. Unmarked fish represented both age-1 and age-2 groups.

DISCUSSION

The low abundance of Rainbow Trout in our sample effort suggested that post-stocking survival is poor in Hayden Lake. We were unable to determine differences in the relative contribution of stocking events and concluded that survival was likely poor for all recent stocking events. Anecdotally, angler reports suggest Rainbow Trout harvest remains low, supporting our observations.

Continuation of Hayden Lake fingerling stocking and subsequent evaluations is recommended despite the apparent poor survival documented in this survey and in prior evaluations. Several variables have been modified recently that may influence our investigations. For example, requested Rainbow Trout fingerling stocking density was increased from 23 to 31 fish/hectare in Hayden Lake in 2017 with an interest in increasing detectability if low to moderate survival is occurring. While a higher stocking rate may be desirable, availability of hatchery product at this time limits options to increase stocking density further. In addition, all-female Kamloops, a strain of Rainbow Trout most similar to strains historically stocked in Hayden Lake that may have demonstrated higher survival, were requested in 2018 and will be stocked in 2019. We recommend continuation of fingerling stocking and evaluation with an understanding that both stocking and evaluation costs are low, and potential return to anglers may be high if a suitable stocking strategy is identified.

Our observed marking rate on kokanee collected from Hayden Lake suggests wild production is low. Subsequently, it did not appear that routine restriction of spawning kokanee into tributaries is necessary at this time. Although we did not detect marks on otoliths from two fish, we had a low level of confidence in our determination that these fish were unmarked. Multiple fish required two structures be processed to clearly identify a thermal mark, suggesting our no-mark detection rate may be less than 100%. As such, our interpretation of observed wild production is supported.

RECOMMENDATIONs

1. Continue to stock and evaluate survival of large (≥ 152 mm) fall fingerling Rainbow Trout stocking efforts by describing relative abundance in Hayden Lake during the spring.

2. Continue periodically monitoring wild production of Hayden Lake kokanee to understand how wild production may impact density-dependent growth.

3. Do not limit kokanee migrations into spawning tributaries at this time as a means of reducing wild production.

Table 28. Date, net set duration (Time (hours)), and location of net sets used to evaluate Rainbow Trout stocking in Hayden Lake, Idaho.

Date	Net	Time	Latitude	Longitude
5/3/2018	1	15.1	47.7573	-116.7229
5/3/2018	2	15.7	47.7527	-116.7213
6/12/2018	3	11.9	47.7805	-116.6824
5/3/2018	4	15.7	47.7503	-116.7557
5/3/2018	5	15.2	47.7496	-116.7009
5/3/2018	6	15.0	47.7509	-116.7036
6/12/2018	7	12.6	47.7724	-116.6820
6/12/2018	8	11.1	47.7838	-116.7011
5/3/2018	9	13.9	47.7669	-116.7413
6/12/2018	10	12.0	47.7805	-116.6741
5/3/2018	11	14.9	47.7478	-116.6948
5/3/2018	12	12.9	47.7664	-116.7473
6/12/2018	13	10.9	47.7685	-116.7236
5/3/2018	14	15.2	47.7527	-116.7109
6/12/2018	15	11.3	47.7876	-116.6898
6/12/2018	16	11.4	47.7739	-116.7077
6/12/2018	17	11.0	47.7720	-116.7096
6/12/2018	18	10.9	47.7671	-116.7178
6/12/2018	19	11.9	47.7795	-116.6870
6/12/2018	20	11.8	47.7817	-116.6931
6/12/2018	21	12.8	47.7581	-116.6923
5/3/2018	22	12.4	47.7639	-116.7527
5/3/2018	23	15.8	47.7538	-116.7230
5/3/2018	24	13.2	47.7667	-116.7439

Table 29. History of Rainbow Trout stocking in Hayden Lake, Idaho from 2011 through 2018. Information provided includes year and season of stocking, hatchery of origin, strain, size, and number of fish stocked, and marks present on stocked fish to identify stocking group.

Year	Period	Hatchery	Strain/Type	Size	Number	Mark
2011	Fall	Grace	Triploid Troutlodge Kamloop	3-6 in. fingerlings	39,600	Ad Clipped
2011	Spring	Nampa	Triploid Troutlodge Kamloop	catchable	472	
2011	Spring	Hagerman	Triploid Troutlodge Kamloop	3-6 in. fingerlings	268,800	
2012	Spring	Grace	Hayspur Rainbow Triploid	3-6 in. fingerlings	18,000	
2012	Spring	Nampa	Triploid Troutlodge Kamloop	catchable	4,832	
2013	Fall	Grace	Hayspur Rainbow Triploid	3-6 in. fingerlings	39,312	
2014	Fall	Cabinet Gorge	Hayspur Rainbow Triploid	3-6 in. fingerlings	38,400	50% Ad Clipped
2015	Fall	Cabinet Gorge	Hayspur Rainbow Triploid	> 6 in. fingerlings	36,520	50% Ad Clipped
2015	Spring	Nampa	Hayspur Rainbow Triploid	catchable	8,867	
2016	Fall	Cabinet Gorge	Unspecified Rainbow Trout	> 6 in. fingerling	25,344	Thermal Marked
2016	Spring	Nampa	Unspecified Rainbow Trout	12 in. catchable	1,535	
2017	Fall	Cabinet Gorge	Unspecified Rainbow Trout	> 6 in. fingerling	50,700	Thermal Marked
2018	Fall	Cabinet Gorge	Troutlodge All Female Kamloop	> 6 in. fingerling	98,601	Thermal Marked

Table 30. Date, water depth, suspended net depth, and location of gill net sets used to evaluate kokanee stocking in Hayden Lake, Idaho.

Date	Net	Water depth (m)	Net depth (m)	Latitude	Longitude
6/19/2018	1	53	9	47.7763	116.70195
6/19/2018	2	53	14	47.7763	116.70195
6/19/2018	3	49	9	47.7763	116.70195
6/19/2018	4	49	15	47.7671	116.70155
6/19/2018	5	50	9	47.7671	116.70155

Table 31. Species sampled, minimum and maximum total length (TL), catch (n), and catch rate (CPUE) from a gill net survey used to evaluate Rainbow Trout stocking in Hayden Lake, Idaho.

Species	Min TL	Max TL	n	CPUE ± 80%CI
Black Crappie	206	256	13	0.57 ± 0.3
Brown Bullhead	274	274	1	0.04 ± 0.1
Hatchery Rainbow Trout	242	242	1	0.04 ± 0.1
Kokanee	228	519	26	1.08 ± 0.5
Northern Pike	765	970	4	0.17 ± 0.1
Tench	375	474	7	0.29 ± 0.2
Westslope Cutthroat Trout	185	185	1	0.04 ± 0.1

BULL TROUT REDD COUNTS

ABSTRACT

In 2018, we counted Bull Trout *Salvenlinus confluentus* redds as an index of adult abundance in two of the major drainages in northern Idaho's Panhandle Region. A total of 53 redds were detected, including 45 redds in the Upper Priest Lake drainage and 8 redds in the St. Joe River drainage. Redd count totals from 2018 were variable relative to average counts from the previous ten-year period, but did not suggest dramatic shifts in Bull Trout abundance at the core area scale. However, continued low redd counts in the St. Joe River drainage suggest that Bull Trout in this core area may be at risk of extirpation.

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INTRODUCTION

Bull Trout *Salvelinus confluentus* were listed by the U.S. Fish and Wildlife Service (USFWS) as a threatened species under the Endangered Species Act in 1998. Thus, monitoring population trends for this species has management importance. Redd counts serve as the primary monitoring tool for Bull Trout populations throughout their range. Idaho Department of Fish and Game (IDFG) personnel, along with employees of other state and federal agencies, annually count Bull Trout redds in standardized stream reaches within each of the four core recovery areas (hereafter core areas) located in the Panhandle Region (USFWS 2015). Redd counts allow for evaluation of the status of populations in these areas and help in directing future management and recovery activities. Results for redd count surveys conducted in tributaries to Lake Pend Oreille are reported separately (Ransom et al. 2020).

METHODS

We counted Bull Trout redds in selected tributaries of the Upper Priest River and St. Joe River drainages where migratory Bull Trout were known or believed to spawn. Bull Trout redd counts were not completed in the Kootenai River drainage in 2018. We located redds visually by walking along standardized sections within each tributary (Ryan et al. 2020c). Surveys were conducted by experienced redd counters or an experienced counter paired with an unexperienced counter in most cases. Unexperienced redd counters were provided basic training in identifying redds prior to a survey. Bull Trout redds were defined as areas of clean gravels at least 0.3 x 0.6 m in size with gravels of at least 76 mm in diameter having been moved by fish and with a mound of loose gravel downstream from a depression (Pratt 1984). In areas where one redd was superimposed over another redd, each distinct depression was counted as one redd. Redd surveys were conducted during a standardized time period (late–September to mid-October). In some surveys, redd locations were recorded on maps and/or recorded by global positioning system (GPS). For reporting purposes, we summarized counts by core area. We compared Bull Trout redd count totals by core area to prior count years to assess long-term trends in redd abundance. Total redd counts were compared to average counts from the previous ten years of sampling. Trends were assessed qualitatively relative to previous count averages rather than by statistical analysis.

RESULTS AND DISCUSSION

Priest Lake Core Area

We completed Priest Lake core area redd counts on September 28, 2018. We counted 45 Bull Trout redds across seven standard stream reaches surveyed in the core area (Table 32). The total redd count represented a decline from the previous year and was below the previous 10-year average for combined counts of 53 redds.

St. Joe Core Area

St. Joe River core area redd counts were completed on September 25–26, 2018. We surveyed three index streams (Wisdom Creek, Medicine Creek, and the mainstem St. Joe River between Heller Creek and St. Joe Lake) that have consistent monitoring data. Additionally, we surveyed Heller Creek, which is not a standardized index stream. We counted a total of four Bull

Trout redds among the three index reaches in the core area (Table 33), all of which were located in Medicine Creek. Four additional redds were counted in Heller Creek. Total redds observed in the index reaches in 2018 represented a continued decline over the past six years, and the total redd abundance remained below the 10-year average for index streams. Of particularly concern, 2018 marked the second consecutive year with only four redds counted in the index streams. This suggests that persistence of this Bull Trout population may be in jeopardy.

The number of streams surveyed per year in the St. Joe River core area has varied considerably over time. However, most spawning is believed to occur in the three index streams that are surveyed annually. Assessment of the Bull Trout population trend should be based on redd counts in the index streams since other streams have been monitored inconsistently. We recommend focusing future efforts primarily on index streams, but surveys of additional streams may be necessary to better assess the conservation status of this population.

RECOMMENDATIONS

1. Continue to monitor Bull Trout spawning escapement through completion of redd surveys.

2. Continue to balance the frequency and location of surveys with the availability of time and intended use of collected data.

3. Engage with U.S. Fish and Wildlife Service staff to discuss possible management actions that may reduce the likelihood of extirpation for Bull Trout in the St. Joe drainage.

Table 32. Bull Trout redd counts by year from the Upper Priest River, Idaho and selected tributaries from 1993 through 2018. Redd surveys were not completed on all stream reaches in all years from 1993 through 2004. As such, mean redd counts for surveys completed in these years may include fewer completed counts.

Stream	Transect Description	Avg. 1993 -2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Upper Priest River	Falls to Rock Cr.	13	5	17	10	36	34	58	25	17	21	16
	Rock Cr. to Lime Cr.	1	2	4	1	0	7	8	12	34	36	12
	Lime Cr. to Snow Cr.*	6	10	3	1	3	6	9	13	11	24	10
	Snow Cr. to Hughes Cr.	4	4	0	7	2	2	0	1	0	4	1
	Hughes Cr. to Priest Lk	0	--	0	0	0	--	--	--	--	--	--
Rock Cr.	Mouth to F.S. trail 308	0.4	0	1	0	0	--	--	--	--	--	--
Lime Cr.	Mouth upstream 1.2 km*	0.2	0	0	0	0	--	--	--	--	--	--
Cedar Cr.	Mouth upstream 3.4 km	0.3	0	0	0	0	--	--	--	--	--	--
Ruby Cr.	Mouth to waterfall	0	0	0	--	--	--	--	--	--	--	--
Hughes Cr.	Trail 311 to trail 312*	1	0	0	0	0	--	--	--	--	--	--
	F.S. road 622 to Trail 311*	1	5	0	7	5	0	3	0	0	0	0
	F.S. road 622 to mouth*	1	3	11	3	2	1	2	3	1	11	1
Bench Cr.	Mouth upstream 1.1 km*	0.3	0	0	0	0	--	--	--	--	--	--
Jackson Cr.	Mouth to F.S. trail 311	0	0	0	0	0	--	--	--	--	--	--
Gold Cr.	Mouth to Culvert*	2	5	6	2	4	3	1	0	0	2	5
Boulder Cr.	Mouth to waterfall	0	0	0	--	0	--	--	--	--	--	--
Trapper Cr.	Mouth upstream 5.0 km upstream from East Fork	2	0	0	--	0	--	--	--	--	--	--
Caribou Cr.	Mouth to old road crossing	0.2	--	0	--	--	--	--	--	--	--	--
All stream reaches combined		29	34	42	31	52	53	81	54	63	98	45

Table 33. Bull Trout redd counts by year from the St. Joe River, Idaho and selected tributaries. Redd surveys were not completed on all stream reaches in all years. As such, mean redd counts for surveys completed between these years may include fewer completed counts.

Stream Name	Avg 1992 - 2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Aspen Cr.	0	--	--	--	--	--	--	--	--	--	--
Bacon Cr.	0	0	--	--	0	--	0	--	--	--	--
Bad Bear Cr.	0	--	--	--	--	--	--	--	--	--	--
Bean Cr.	7	1	--	--	1	0	--	--	--	--	--
North Fork Bean Creek	--	--	--	--	19	8	0	--	--	--	--
Unnamed tributary to N.Fk. Bean	--	--	--	--		3	--	--	--	--	--
Beaver Cr.	0	0	3	--	0	--	--	--	--	--	--
Bluff Cr.- East Fork	0	--	--	--	--	--	--	--	--	--	--
California Cr.	1	2	--	--	0	--	--	0	--	--	--
Cascade Creek	--	--	--	--	2	--	--	--	--	--	--
Copper Cr.	0	--	--	--	--	--	--	--	--	--	--
Entente Cr.	0	--	--	--	--	--	--	--	--	--	--
Fly Cr.	1	1	0	--	0	--	--	3	--	--	--
Gold Cr. Lower mile	0	--	--	--	--	--	--	--	--	--	--
Gold Cr. Middle	0	--	--	--	--	--	--	--	--	--	--
Gold Cr. Upper	1	--	--	--	--	--	--	--	--	--	--
Gold Cr. All	0	--	--	--	--	--	--	--	--	--	--
Heller Cr.	1	3	9	5	5	--	0	11	--	5	4
Indian Cr.	0	--	--	--	--	--	--	--	--	--	--
Medicine Cr.	38	41	48	35	20	20	17	4	11	3	4
Mill Cr.	--	--	--	--	9	6	--	--	--	--	--
Mosquito Cr.	1	--	--	--	--	--	--	--	--	--	--
My Cr.	--	--	--	--	0	--	--	--	--	--	--
Pole	--	--	--	--	0	--	--	--	--	--	--
Quartz Cr.	0	--	--	--	--	--	--	--	--	--	--
Red Ives Cr.	0	--	2	4	0	--	0	0	--	0	--
Ruby Cr.	3	--	--	--	0	--	--	--	--	--	--
Sherlock Cr.	1	--	1	--	2	--	0	0	--	--	--
Simmons Cr. - Lower	0	0	--	--	--	--	--	--	--	--	--
Simmons Cr. - NF to Three Lakes	2	0	--	--	--	--	--	--	--	--	--
Simmons Cr. - Three Lakes to Rd 1278	2	0	--	--	--	--	--	--	--	--	--

Table 33 (continued)

Stream Name	Avg 1992 - 2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Simmons Cr. - Rd 1278 to Washout	0	0	--	--	--	--	--	--	--	--	--
Simmons Cr. - Upstream of Washout	0	0	--	--	--	--	--	--	--	--	--
Simmons Cr. - East Fork	0	0	--	--	--	--	--	--	--	--	--
St. Joe River - below Tento Creek	0	--	--	--	--	--	--	--	--	--	--
St. Joe River - Spruce Tree CG to St. J. Lodge	0	--	--	--	--	--	--	--	--	--	--
St. Joe River - St. Joe Lodge to Broken Leg	4	--	--	--	--	--	--	--	--	--	--
St. Joe River - Broken Leg Cr upstream	0	--	--	--	--	--	--	--	--	--	--
St. Joe River - Bean to Heller Cr.	0	--	--	--	--	--	--	--	--	--	--
St. Joe River - Heller to St. Joe Lake	8	1	5	7	4	1	0	7	2	1	0
Three Lakes Creek	0	--	--	--	--	--	--	--	--	--	--
Timber Cr.	0	--	--	--	--	--	--	--	--	--	--
Tinear Cr.	--	--	--	--	2	5	--	--	--	--	--
Wampus cr	0	--	--	--	--	--	--	--	--	--	--
Washout cr.	1	--	--	--	--	--	--	--	--	--	--
Wisdom Cr	10	8	1	1	5	1	0	0	0	0	0
Yankee Bar	0	--	--	--	--	--	--	1	--	--	--
Total - Index Streams	55	50	54	43	29	22	17	11	13	4	4
Total - All Streams	63	57	69	52	69	44	17	26	13	9	8
Number of streams counted	14	15	8	5	18	8	7	9	3	5	4

COEUR D'ALENE LATERAL LAKES NORTHERN PIKE POPULATION EVALUATIONS

ABSTRACT

Northern Pike populations were sampled in the "Chain Lakes" area along the Coeur d'Alene River during 2016–2018 to characterize population structure and dynamics, and to establish baseline information to guide management decisions. Population relative abundance was not as high as expected, and comparatively low to subpopulations occupying good habitat (i.e., bays) in Lake Coeur d'Alene. Relative abundance varied from 0.4–1.0 fish/net h among lakes. Medicine Lake exhibited the highest mean and variance about relative abundance. Size structure followed general expectations with most populations characterized by few preferred length and larger individuals. Mean total length varied from 582 to 705 mm and mean length surprisingly showed relatively little variability within populations. Total annual mortality varied widely (26–83%), but estimates were generally high compared to studies across the distribution of Northern Pike. Fishing was not an important mortality component in most lakes, with the exception of Anderson and Thompson lakes. Annual angler exploitation varied from zero to 33% among lakes and mean estimated exploitation was 14%. Mean exploitation was less than 50% of previous estimates for Northern Pike populations in Coeur d'Alene Lake. Northern Pike growth in the Chain Lakes is generally slow, especially in comparison to populations in larger systems such as Hayden and Coeur d'Alene lakes. We interpret our results to mean that Northern Pike management actions beyond the current approach are likely not necessary. Northern Pike occur at lower abundances in the Chain Lakes that previously thought, but appear to not be strongly influenced by fishing mortality in some systems. This understanding is contrary to what has been hypothesized to occur in most Panhandle Region Northern Pike fisheries, and based on previous information is not, in fact, congruent with relationships between abundance and angler interactions in the rest of the Coeur d'Alene Lake system.

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INTRODUCTION

Northern Pike *Esox lucius* were illegally introduced into the Chain Lakes along the Lower Coeur d'Alene River sometime during the early 1970s (Rich 1992). It is thought that Northern Pike were initially stocked in Cave Lake (Rich 1992) and subsequently expanded into the other nearby lateral lakes and downstream into Lake Coeur d'Alene. Populations of Northern Pike can now be found throughout the Lake Coeur d'Alene system downstream of Cataldo, Idaho on the Coeur d'Alene River and essentially downstream of St. Maries, Idaho in the St. Joe River drainage. Northern Pike have been illegally transferred to other lowland lakes around the Panhandle Region where sustained populations can now be found. Currently, the known distribution of Northern Pike in Idaho is relegated to Idaho's five northern counties. It is widely recognized that Northern Pike have strong potential to alter fish communities and negatively influence populations of native and nonnative sport fishes. However, Northern Pike also support some of the Panhandle Region's most popular fisheries and, as it stands, the spring Northern Pike fishery in the Lake Coeur d'Alene system is one of the most significant for this species in the Pacific Northwest.

Northern Pike are formally recognized as a game fish by the state of Idaho and management policy focuses on limiting their distribution to its current extent. State management policy seeks to achieve this objective by discouraging the transfer of Northern Pike to water bodies outside of their current distribution by mandating unlimited harvest rules without size restrictions and prohibiting catch-and-release tournament events. The overarching intent of current policy is to recognize the value of Northern Pike populations to the angling public while simultaneously managing those populations liberally to limit their abundance with angling.

Northern Pike populations in northern Idaho are probably controlled by a combination of angler exploitation and environmental conditions. Most importantly, high angler exploitation rates are hypothesized to limit Northern Pike populations to the extent that negative interactions with existing fish communities is minimal. Previous estimates of angler exploitation have been high and relative fish densities low (Rich 1992; Walrath 2013). However, contemporary estimates of angler exploitation and descriptions of population characteristics are lacking in many systems, especially the Chain Lakes. As such, information gaps have limited IDFG's ability to assess the efficacy of its State Fishery Management Plan relative to Northern Pike and ensure a diversity of warmwater angling opportunities.

During 2016–2018, we sampled Northern Pike populations in seven of the Chain Lakes. Following a study conducted in Killarney Lake during 2014 (Watkins et al. 2018), we sought to describe the structure of Northern Pike populations and evaluate how anglers interact in the fisheries supported by Northern Pike. As such, our study was intended to compliment that of Watkins et al. (2018) and provide a holistic assessment of Northern Pike in the Chain Lakes area.

OBJECTIVES

1. Describe population dynamics (i.e., growth, mortality) and structure.

2. Estimate angler exploitation of Northern Pike.

3. Describe the presence of various species co-occurring with Northern Pike.

STUDY AREA

The Coeur d'Alene River lateral lakes or "Chain Lakes" are lentic water bodies situated in the floodplain along the lower Coeur d'Alene River between its confluence with Lake Coeur d'Alene and Rose Lake, Idaho. Historically, the Chain Lakes were seasonally flooded by the Coeur d'Alene River and disconnected from the main channel during base follow. During Captain John Mullan, Junior's exploration of the Coeur d'Alene River corridor, he noted vast wetland areas in the lower river floodplain, punctuated by a series of distinct lakes. The Chain Lakes are essentially bounded by the Coeur d'Alene Mountains along the northern or southern end and a series of artificial dykes. The lower Coeur d'Alene River has undergone substantial alteration since this time to include dyking and channel development between the river and the Chain Lakes. These alterations have resulted in the understanding that the Chain Lakes developed in relatively recent history following river embankment development that "trapped" laterally moving water in floodplain depressions and deep wetlands. However, with respect to these environments, Captain Mullan's account and observations of the Coeur d'Alene River floodplain indicate that the area was likely very similar to what it is today with the exception of the temporally continuous connectivity within the system.

Each of the Chain Lakes are relatively similar in surface area (range = 233 ha), but differ more in terms of bathometry, catchment size, and public accessibility. Some of the lakes possess shoreline access or direct boat access while others are accessible via overwater boat travel only. All of the Chain Lakes support warmwater fish assemblages throughout the year and coldwater species seasonally. Fish assemblage sampling has been infrequent with the exception of standard lowland lake surveys on Medicine, Swan, and Anderson lakes in 1981, Swan and Black lakes in 1995, Anderson Lake in 1996, Killarney and Cave lakes in 1998, and Killarney and Black lakes in 2008.

METHODS

Sampling was conducted during early-April through early-May of each year to coincide with Northern Pike spawning. Time and staffing constraints prohibited sampling of all lakes during the same year. As such, we sampled 2–3 lakes per year over the course of the study. A simple random sampling design was used to allocate effort to various 400-m long shoreline units in each lake. Sinking experimental gill nets (45 × 1.8 m; 5 panels with 50-, 64-, 76-, 88-, and 100-mm stretch-measure mesh) were used to capture fish. A single gill net was deployed perpendicular to the shoreline in each unit and fished for approximately 1–3 hours (mean = 2.2 h) to minimize capture mortality of Northern Pike.

Catch-per-unit-effort (CPUE) was summarized as the number of fish sampled per net/h and averaged among all deployments. Total length (TL; mm) was measured from all fishes and used to inform our understanding of Northern Pike population size structure. Two to three leading fin rays were removed from the pelvic fin of each fish for age estimation. Fin rays were allowed to air dry and subsequently mounted in epoxy using 2 mL microcentrifuge tubes following Koch and Quist (2007). Cross sections (0.9 mm thick) were cut near the base of each dorsal spine just distal to the articulating process using an Isomet low-speed saw (Buehler Inc., Lake Bluff, Illinois, USA). Resulting dorsal spine cross-sections were viewed using a dissecting microscope with transmitted light and an image analysis system (Image ProPlus; Media Cybernetics, Silver Springs, Maryland, USA). Annuli were enumerated on all structures by a single reader. Knowledge of biological information for each fish was unknown during the age estimation process to avoid bias.

Age structure of Northern Pike was summarized for each population. Total annual mortality (*A*) was estimated using a weighted catch curve (Miranda and Bettoli 2007). Northern Pike were typically fully-recruited to the sampling gear at either age-2 or 3, so *A* was only estimated for fish older than 2–3 years of age depending on age-at-recruitment. Mean length at age information was summarized and a von Bertalanffy growth function (von Bertalanffy 1938) was fitted to those data to assess patterns in growth.

Angler exploitation of Northern Pike was evaluated using tag return information. Fish were fitted with an orange, non-reward FD-94 T-bar anchor tag (76 mm; Floy Tag Inc., Seattle Washington, USA) after processing for biological information and released. Tags were uniquely numbered and inserted near the posterior end of the dorsal fin of each Fish. All tags also possessed the telephone number and web address for IDFG's "Tag! You're It!" reporting hotline. Angler exploitation was estimated using the non-reward tag reporting estimator described by Meyer et al. (2012), namely,

$$\mu' = \mu / [\lambda\ (1–Tag_l)(1–Tag_m)]$$

where μ' is the adjusted angler exploitation rate, μ is the unadjusted exploitation rate (i.e., number of fish reported divided by the number of fish tagged), λ is the species-specific angler reporting rate (53.0%), Tag_l is the tag loss rate (10.2%), and Tag_m is the tagging mortality rate (3.0%). Annual angler exploitation rates were estimated for each lake following one year at-large.

RESULTS

Relative abundance of Northern Pike populations varied from 0.4–1.0 fish/net h among lakes (Table 34; Figure 33). The highest variance about mean CPUE occurred in Medicine Lake but, otherwise, populations exhibited very similar variance about our estimates. Fish communities in each lake were relatively similar and generally composed of simple warmwater assemblages (Table 35). Size structure of Northern Pike followed general expectations with most populations characterized by few preferred length and larger individuals. Mean total length varied from 582 to 705 mm, and mean length surprisingly showed relatively little variability within populations. Overall, size distributions did not show patterns or major dissimilarities among lakes (Figure 34). Age estimates varied from 1 to 8 years among populations, but old (i.e., age-5+) individuals were poorly represented in all of the study populations. Total annual mortality varied widely (26–83%) and was not closely associated with estimates of harvest. Northern pike recruited to sinking gill nets at age-3 in most populations. Annual angler exploitation varied from zero to 33% among lakes and mean combined exploitation was 14% (Table 34; Figure 35). Mean exploitation was less than 50% of previous estimates for Northern Pike populations in Coeur d'Alene Lake (Walrath 2013).

Northern Pike growth in the Chain Lakes is generally slow, especially in comparison to populations in larger lakes such as Hayden and Coeur d'Alene lakes (Rich 1992; Walrath 2013; Ryan et al. 2020a). Estimates of von Bertalanffy growth model parameters varied widely and did not show discernable patterns among populations. Theoretical maximum length estimates were typically below 800 mm and growth coefficient estimates were lower than other northern Idaho populations, suggesting that early life history growth rate was slow relative to nearby populations.

DISCUSSION

Northern Pike management has become an increasingly important focus for fishery professionals in the Pacific Northwest because of the threat the species can pose to existing fish communities. Fish and wildlife management agencies deal with a host of issues relative to nonnative piscvores, such as Northern Pike, but it is widely understood that the magnitude and importance of those issues are fishery- and system-specific. For example, in Lake Coeur d'Alene, Northern Pike occur at low density due in part to the patchiness of suitable habitat and high rate of angler harvest. In most Northern Pike fisheries, particularly those occurring in systems with abundant habitat, it has been thought that high angler exploitation regulates Northern Pike abundance to a large degree. However, our results differed substantially relative to our expectations about how Northern Pike populations are structured and how anglers interact with populations in the Chain Lakes. Indeed, Northern Pike angling appears to be an important spring fishery throughout the Chain Lakes, but we estimated lower than expected rates of exploitation and relative abundances that were variable and not closely associated among lakes.

Although each of the Chain Lakes provide suitable Northern Pike habitat, relative abundances were similar to what has been documented in bays throughout Lake Coeur d'Alene (Rich 1992; Walrath 2013). Walrath (2013) commented that high abundance Northern Pike populations are generally characterized by catch rates of 1.0 fish/net h and greater across the species' distribution, and Paukert and Willis (2003) provided a similar suggestion based on information from Nebraska pothole lakes populations. In comparison to the populations in our study, Chain Lakes Northern Pike populations are at moderate abundance in most lakes (Figure 33). The exception is Medicine Lake which exhibited CPUE approaching 1.0 fish/net h.

In general, Northern Pike populations in the Chain Lakes were characterized by fast growth, low longevity, poor size structure, and high mortality. Chain Lakes populations exhibited somatic growth patterns similar to Lake Coeur d'Alene, but maximum length in the Chain Lakes was consistently low and very different than Lake Coeur d'Alene populations in that regard. Rich (1992) and Walrath (2013) reported good size structure of Lake Coeur d'Alene subpopulations and routinely sampled fish exceeding 1,000 mm TL. We did not sample any fish over 1,000 mm TL and estimated mean and maximum TLs well below that reported by Walrath (2013). We speculate that the Chain Lakes are probably forage limited and not capable of supporting high abundances of Northern Pike, particularly fish of large size that have high metabolic demands. Of course, length structure is often associated with fish age, and in our study populations maximum age rarely exceeded five years. Contrasting this information with Walrath (2013), longevity of Northern Pike in Lake Coeur d'Alene was typically 8–9 years in bay habitats. Forage limitations for adult Northern Pike likely result in direct mortality of older age-classes or emigration of large individuals seeking better habitat. Combining information about Northern Pike abundance and age and size structure provides some understanding about the carrying capacity and production potential of the Chain Lakes relative to Northern Pike. While our study lakes support moderate abundance relative to the Panhandle Region, the potential to maintain moderate abundance of populations also having large, high demand individuals is low.

Fishing was not always an important component of Northern Pike mortality in most of our study systems. Annual exploitation was between 10% and 20% for the majority of the study lakes and was generally lower than estimates for bays in Lake Coeur d'Alene and Benewah Lake (Walrath 2013; Figure 35). Angler exploitation estimates for Thompson and Medicine lakes were the only estimates that approached values common for the greater Lake Coeur d'Alene system. This pattern is intuitive considering the variability in angler access throughout the basin. That is, most places with adequate access for shoreline and boat anglers tended to exhibit relatively high harvest (e.g., 20–30%), whereas lakes with more difficult access (particularly lack of access for bank anglers) had much lower harvest rates (i.e., 0–15%).

RECOMMENDATIONS

1. Maintain liberal angling rules for Northern Pike in the Chain Lakes.

2. Periodically monitor Northern Pike populations and fish community assembly in the Chain Lakes to assess changes and potential interspecific interactions.

Table 34. Sample size (n), mean catch-per-unit-effort (CPUE = fish/net h), total length (mm; Minimum–Maximum [Min–Max]) statistics, total annual mortality (A), annual angler exploitation (μ), and von Bertalanffy growth function (VBGM) for Northern Pike populations sampled from the Coeur d'Alene lateral lakes in northern Idaho (2016–2018). Numbers in parentheses represent one standard error of the mean.

Water body	n	CPUE	Total length		A	µ	VBGM
			Mean	Min–Max			
Anderson Lake	22	0.5 (0.2)	705.1 (20.8)	577–953	57.5	0.20	$L_{age} = 824(1 - e^{(-0.675(age - 0.30))})$
Black Lake	15	0.4 (0.1)	651.9 (47.2)	384–991	26.1	0.00	$L_{age} = 869(1 - e^{(-0.643(age - 0.903))})$
Blue Lake	56	0.7 (0.2)	618.0 (10.9)	439–883	83.0	0.10	$L_{age} = 805(1 - e^{(0.48(age + 0.100))})$
Cave Lake	21	0.5 (0.2)	626.1 (26.2)	329–831	62.2	0.10	$L_{age} = 758(1 - e^{(-0.724(age + 0.247))})$
Medicine Lake	38	1.0 (0.3)	582.6 (23.7)	265–805	53.0	0.15	$L_{age} = 746(1 - e^{(-0.667(age - 0.398))})$
Swan Lake	46	0.6 (0.1)	582.8 (11.4)	400–775	58.1	0.13	$L_{age} = 710(1 - e^{(-0.54(age + 0.152))})$
Thompson Lake	20	0.4 (0.1)	670.5 (18.0)	478–780	55.3	0.33	$L_{age} = 769(1 - e^{(0.425(age + 0.10))})$

Table 35. Fishes sampled from the Coeur d'Alene lateral lakes in northern Idaho as part of Northern Pike population assessment during 2016–2018. Taxa include Brown Bullhead (BBH), Black Crappie (BCR), Bluegill (BLG), Chinook Salmon (CHK), Largemouth Bass (LMB), Largescale Sucker (LSS), Northern Pike (NPK), Northern Pikeminnow (NPM), Pumpkinseed (PKS), Smallmouth Bass (SMB), Tench (TNC), Westslope Cutthroat Trout (WCT), and Yellow Perch (YLP).

Water body	BBH	BCR	BLG	CHK	LMB	LSS	NPK	NPM	PKS	SMB	TNC	WCT	YLP
Anderson Lake	X	X	X	X	X	X	X		X		X		X
Black Lake	X	X			X	X	X	X		X	X		X
Blue Lake	X				X	X	X			X		X	X
Cave Lake	X	X			X	X	X	X			X	X	X
Medicine Lake	X	X			X	X	X				X		X
Swan Lake	X	X	X		X		X			X	X		X
Thompson Lake	X	X	X		X	X	X		X	X	X		X

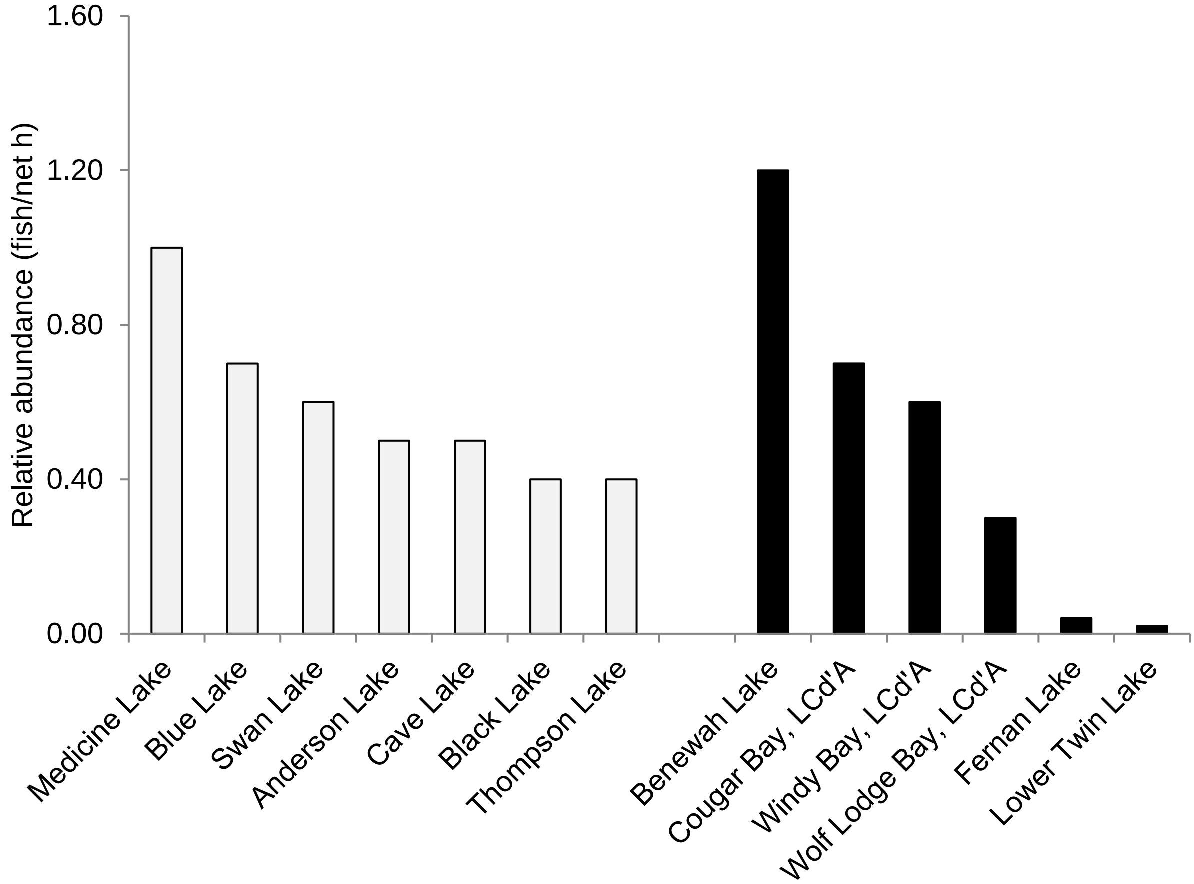


Figure 33. Comparison of relative abundance among Northern Pike populations in the “Chain Lakes” (gray bars) and other waters throughout the Panhandle Region (black bars; LCd’A = Lake Coeur d’Alene).

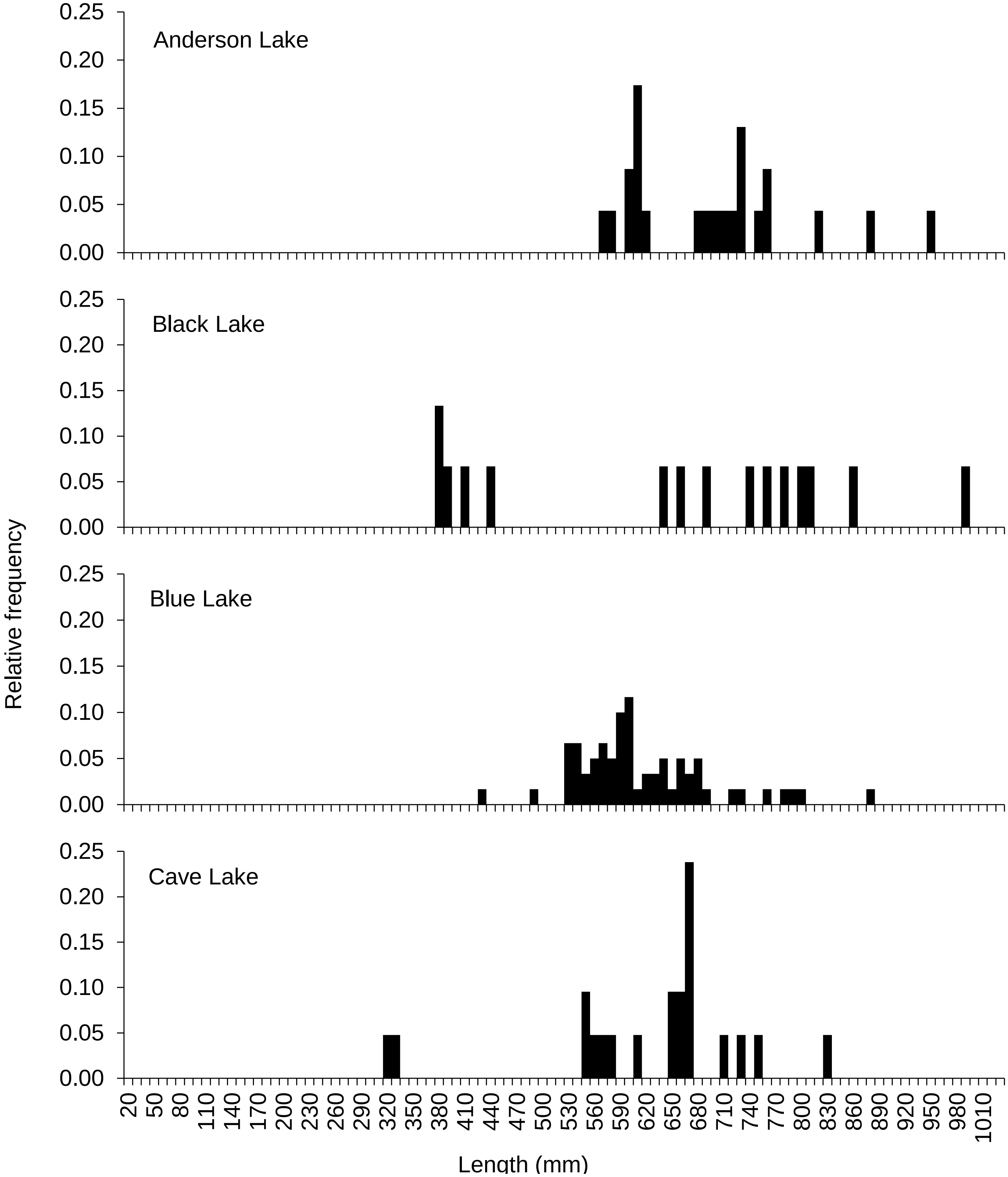
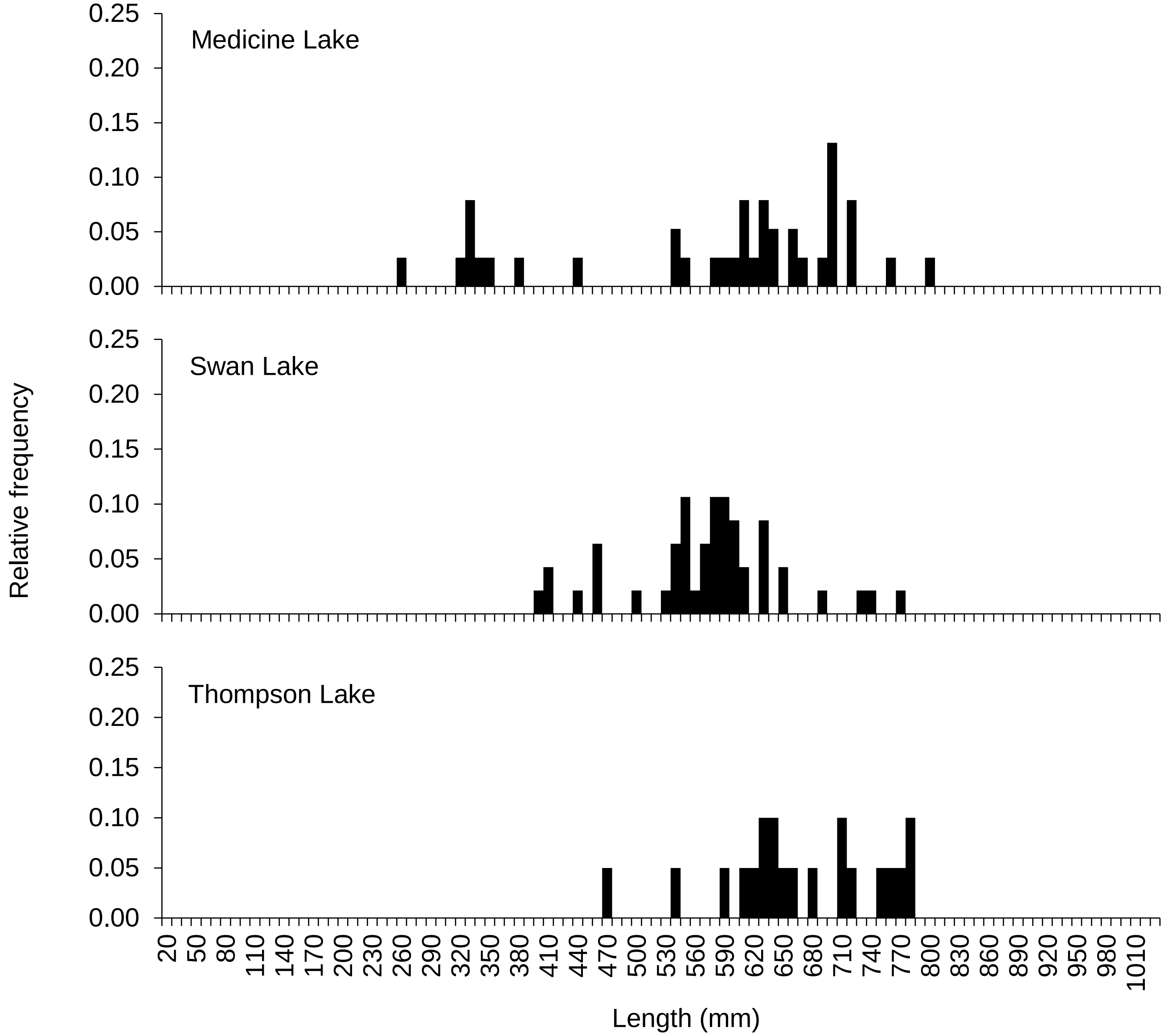


Figure 34. Length-frequency distributions for Northern Pike populations sampled from the Coeur d'Alene River "Chain Lakes" (2016–2018).

Figure 34 (continued)



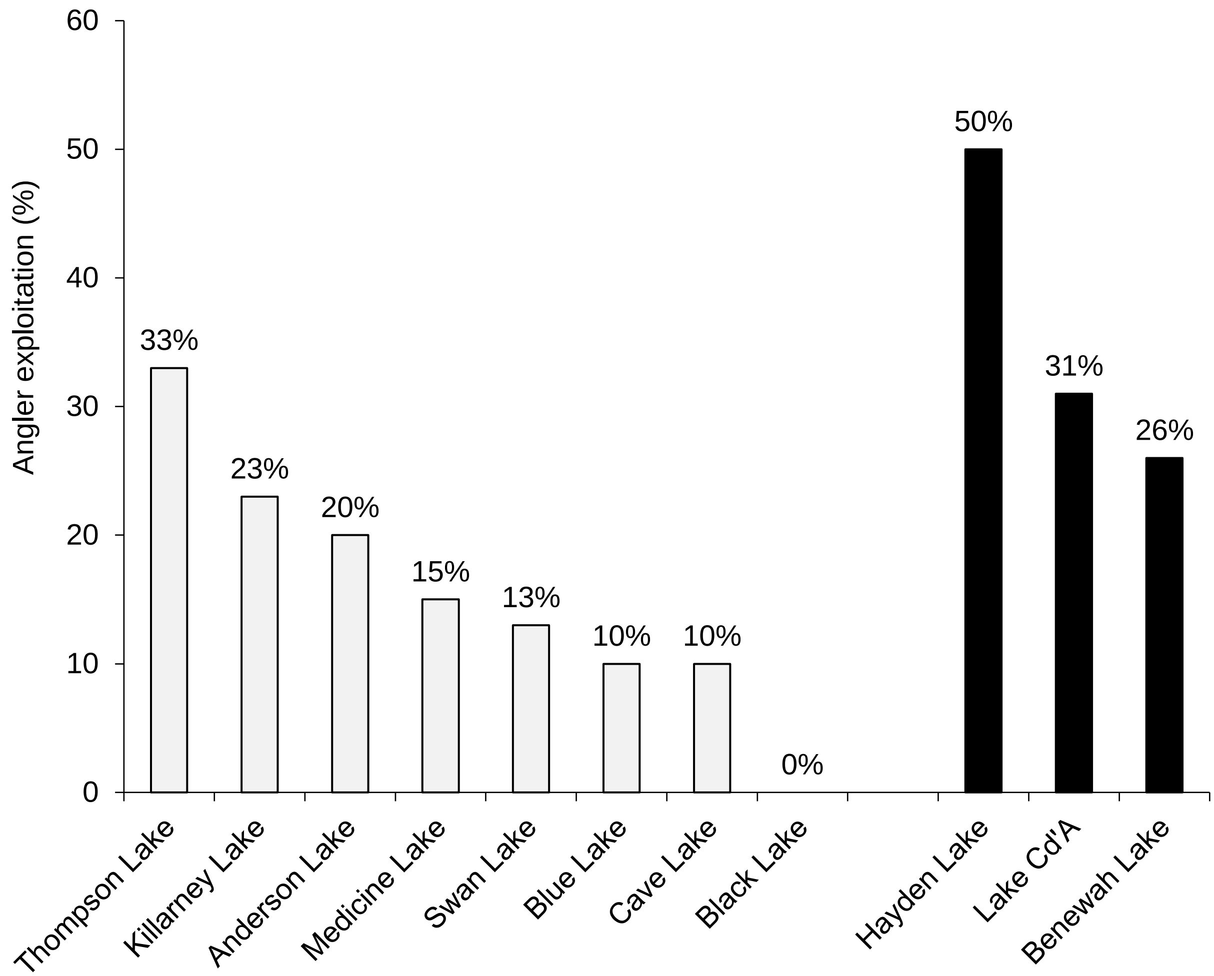


Figure 35. Comparison of angler exploitation rates for the “Chain Lakes” (gray bars) and other nearby Panhandle Region Northern Pike fisheries (black bars; Lake Cd’A = Lake Coeur d’Alene).

LAKE COEUR D’ALENE AND SPIRIT LAKE KOKANEE EVALUATIONS

ABSTRACT

We estimated age-specific abundance, density, and population characteristics of kokanee *Oncorhynchus nerka* in Lake Coeur d’Alene and Spirit Lake to monitor population trends. A modified midwater trawl was used to sample kokanee during July 6–8, 2018. We estimated a total abundance of 1,993,211 and 311,506 kokanee in Lake Coeur d’Alene and Spirit Lake, respectively. The Lake Coeur d’Alene kokanee population had somewhat below average abundance of adult fish during 2018, but the relatively low abundance of age-1 and 2 fish confirmed the presence of weak year-classes in 2016 and 2017. We also documented weak 2016 and 2017 year-classes in Spirit lake; however, total abundance in Spirit Lake has been low relative to our most recent surveys. Mean total length of adult kokanee in Lake Coeur d’Alene was 283 mm, which meets the longstanding management objective. We again documented below average adult kokanee densities in Spirit Lake, suggesting that several years of consecutively low recruitment and high adult mortality have manifested in the fishery. Size structure of kokanee in Spirit Lake was better than in previous years (mean age-3 TL = 243 mm) and growth improved. Recruitment during 2014–2016 was relatively low, suggesting that the trends in growth, and subsequently size structure, may continue to improve. However, recruitment was strong again in 2017 and 2018. We recommend continued monitoring of both kokanee populations to assess trends in age-specific abundance and growth. Monitoring should focus on assessing the fishery-level effects of in both lakes from recent weak year-classes.

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INTRODUCTION

Kokanee *Oncorhynchus nerka* are a popular sport fish across much of the western U.S. because of their high catchability and table value. Kokanee angling is especially popular among local anglers because it is family-oriented, consistently entertaining, and requires simple gear. Kokanee comprise much of the fishing effort in northern Idaho lakes, making them an important focus for management. The Idaho Department of Fish and Game's (IDFG) current policy is to manage for adult kokanee abundances that support high annual harvest yields and provide prey for predators. Current and continued evaluations of kokanee populations in Lake Coeur d'Alene and Spirit Lake will provide information necessary to manage these fisheries.

Kokanee were introduced to Lake Coeur d'Alene in 1937 by the IDFG to establish a harvest-oriented fishery (Goodnight and Mauser 1978; Hassemer and Rieman 1981; Ryan et al. 2014). Initial introductions were made from a late-spawning shoreline stock from Lake Pend Oreille (originally Lake Whatcom, WA stock). During the early 1970s, attempts were made to introduce kokanee from an early-spawning stock (Meadow Creek, British Columbia) into Lake Coeur d'Alene; however, early-spawning kokanee failed to establish a wild population and had dwindled by 1981 (Goodnight and Mauser 1980; Mauser and Horner 1982). Despite unsuccessful attempts to establish early-spawners, the kokanee fishery peaked in the mid-1970s and the wild, late-run stock was producing annual yields between 250,000–578,000 fish during that time (Goodnight and Mauser 1976; Goodnight and Mauser 1980; Rieman and LaBolle 1980). By the early 1980s, fishery managers had documented density-dependent effects on adult size structure of kokanee which prompted an increase in the daily bag limit from 25 to 50 fish per day and the introduction of Chinook Salmon *O. tshawytscha* as a biomanipulation tool to reduce kokanee abundance (Mauser and Horner 1982). Chinook Salmon naturalized in the system and are now an important component of the Lake Coeur d'Alene fishery. In recent history, the kokanee population has not been highly influenced by abundance of predators, but rather by environmental conditions, particularly spring flooding.

Kokanee populations are greatly influenced by environmental conditions. For example, stochastic natural events can alter dynamic rate functions and have long-lasting effects on a population (Hassemer 1984). Poor recruitment commonly results from adverse environmental conditions and can be problematic from a fisheries management standpoint because kokanee are semelparous, and thus it may take several generations for recruitment to return to form. This dynamic was shown in Lake Coeur d'Alene where weak year-classes have resulted from high spring runoff events (i.e., 1996 flooding). The weak 1996 year-class resulted in low recruitment during subsequent years and translated into low abundance of harvestable age-3 and age-4 kokanee during 1998–2003. Lake Coeur d'Alene supports several predator species which prey upon kokanee at various life stages. As such, poor environmental conditions coupled with high predator abundance can have cumulative negative effects on kokanee dynamic rate functions, and thus abundance. The IDFG maintains long-term data on kokanee population dynamics and abundance in Lake Coeur d'Alene to continually evaluate population-level changes resulting from environmental factors and fishery management. In addition, annual assessment of the kokanee population provides IDFG with valuable information that can be provided to anglers.

Late-spawning kokanee were also transplanted from Lake Pend Oreille to Spirit Lake in the late-1930s (Ryan et al. 2014), and this stock has essentially supported the wild component of the fishery. According to Rieman and Meyers (1990), Spirit Lake historically produced some of the highest relative annual yields of kokanee throughout the western U.S. and Canada. Attempts have been made to establish early-spawning kokanee to diversify the fishery, the last being in 2008 (Maiolie and Fredericks 2013). However, it has been thought that beaver dams and limited spawning habitat precluded them from naturalizing and significantly contributing to the fishery. Recent population assessments have shown that abundance of wild late-spawning adults has been high, so stocking was discontinued in 2010. In fact, recent kokanee assessments have shown fish are exhibiting slow growth relative to other systems, likely due to density-dependent effects.

OBJECTIVES

1. Maintain long-term monitoring data to provide information related to kokanee management in Lake Coeur d'Alene and Spirit Lake.

2. Estimate abundance and describe population characteristics of kokanee populations in Lake Coeur d'Alene and Spirit Lake.

STUDY AREA

Lake Coeur d'Alene

Lake Coeur d'Alene is a mesotrophic natural lake located in the Panhandle of northern Idaho (Figure 36). Lake Coeur d'Alene lies within Kootenai and Benewah Counties and it is the second largest natural lake in Idaho with a surface area of 12,742 ha, mean depth of 24 m, and maximum depth of 61 m (Rich 1992). The Coeur d'Alene and St. Joe rivers are the major tributaries to Lake Coeur d'Alene; however, many smaller tributaries also exist. The outlet to Lake Coeur d'Alene is the Spokane River, a major tributary to the Columbia River. Water resource development in the lake includes Post Falls Dam which was constructed on the Spokane River in 1906, and raised the water level approximately 2.5 m. In addition to creating more littoral habitat and shallow-water areas, the increased water level created more pelagic habitat for pelagic salmonids (e.g., kokanee, Chinook Salmon).

The fishery in Lake Coeur d'Alene can be broadly characterized as belonging to one of three components—kokanee, Chinook Salmon, or warmwater species; all of which are popular among anglers. The fish assemblage has become increasingly complex over time, particularly during the past 30 years. Increased fish assemblage complexity has undoubtedly resulted in increased biological interactions, but also diversified angler opportunity. Because of its close proximity to several major cities (i.e., Coeur d'Alene, Spokane), Lake Coeur d'Alene generates high angling effort, contributing considerably to state and local economies.

Spirit Lake

Spirit Lake is located in Kootenai County near the town of Spirit Lake, Idaho (Figure 37). The lake has a surface area of 596 ha, a mean depth of 11.4 m, and a maximum depth of 30.0 m. Brickel Creek is the largest tributary to the lake and drains a forested interstate watershed extending into eastern Washington. Brickel Creek originates on the eastern slope of Mount Spokane at approximately 744 m in elevation and flows in an easterly direction before forming Spirit Lake. Spirit Lake discharges into Spirit Creek, an intermittent outlet located at the northeastern end of the lake; Spirit Creek flows into the Rathdrum Prairie where flow typically becomes subterraneous and contributes to the Rathdrum Aquifer. Spirit Lake is considered mesotrophic having the following water quality concentrations: chlorophyll *a* = 5.3 µg/L (Soltero and Hall 1984), total phosphorus = 18 µg/L, and Secchi depth = 3.9 m (Rieman and Meyers 1992).

The fishery in Spirit Lake has three main components—kokanee, Westslope Cutthroat Trout (stocked as fingerlings), and warmwater species. Size structure of kokanee in Spirit Lake has been poor in recent years and anglers have generally lost interest in the fishery. When conditions allow, the lake supports a popular ice fishery targeting kokanee and Yellow Perch *Perca flavescens*.

METHODS

Population Monitoring

During 2018, kokanee were sampled from Spirit Lake and Lake Coeur d'Alene on July 8 and 6–7, respectively. Kokanee were sampled using a modified midwater trawl (hereafter referred to as the trawl) towed by a 9.2-m boat at a speed of 1.55 m/s. The trawl has been successfully employed in large lentic systems for sampling kokanee (Rieman 1992). The trawl consisted of a fixed frame (3.2 m × 2.0 m) and a single-chamber mesh net (6.0-mm delta-style No. 7 multifilament nylon twine, knotless mesh). Further, the trawl assembly consists of two winch-bound cable towlines which are each passed through a single pulley block. The pulley blocks are vertically-attached to a 2.4 m-tall frame mounted to the stern of the boat allowing the trawl to be easily deployed and retrieved during sampling. Further information on the trawl can be found in Bowler et al. (1979), Rieman (1992), and Maiolie et al. (2004).

Trawling was conducted at 21 and 5 predetermined transects throughout Lake Coeur d'Alene and Spirit Lake, respectively (Figure 36; Figure 37). Transects were originally assigned using a systematic sampling design within three arbitrary strata (i.e., Sections 1, 2, and 3) and have remained the same to standardize abundance estimates (Ryan et al. 2014). During fish sampling, the bottom and top of the kokanee layer was identified using the onboard sonar unit, and the trawl was towed in a stepwise pattern (2.4-m increments; three minutes per step) to capture the entire layer at each transect (Rieman 1992). Upon retrieval of the trawl, kokanee were measured for total length (TL; mm) and saggital otoliths were collected from 10 individuals per 1-cm length group if available. Otoliths were removed following the procedure outlined by Schneidervin and Hubert (1986) and horizontally mounted in epoxy using PELCO flat embedding molds (Ted Pella, Inc., Redding, California, USA). Otoliths were cross-sectioned transversely with sections bracketing the nucleus to capture early annuli. Resulting cross-sections were polished with 1,000 grit sandpaper and viewed using a dissecting microscope to estimate age.

Lake Coeur d'Alene Spawner Assessment

Kokanee spawner length and age structure was estimated to evaluate growth objectives. Mature adults were sampled during December 7, 2018 using a sinking experimental gill net (46.0 m × 1.8 m with panels of 50-, 64-, 76-, 88-, and 100-mm stretch-measure mesh). Gillnets were fished overnight in the vicinity of Higgens Point in Wolf Lodge Bay where kokanee index netting has historically occurred. Sampled fishes were sexed and measured for TL (mm). In addition, otoliths were removed from five individuals per 1-cm length group immediately after sampling. Whole otoliths were viewed by a single reader using a dissecting microscope with reflected light to estimate age.

Data Analysis

Age structure of both populations and Lake Coeur d'Alene spawners was estimated using an age-length key (Isermann and Knight 2005; Quist et al. 2012). Age data was then used to generate estimates of age-specific abundance. Total population abundance estimates have traditionally been used to index the kokanee populations in both Spirit and Coeur d'Alene lakes. Therefore, we calculated total age-specific abundance (*N*) which could be compared to prior surveys. Length-frequency information from trawling and spawner index netting was analyzed to provide insight on size structure and length-at-age.

RESULTS

Lake Coeur d'Alene Population Monitoring

We sampled a total of 797 kokanee by trawling in Lake Coeur d'Alene. We estimated a total population abundance of 1,993,211 kokanee and density of 181 kokanee/ha. Age-specific abundance was estimated in order to make prior year comparisons and to provide insight on recruitment of adults to the fishery. We estimated abundances of approximately 1 million age-0, 503,000 age-1, 58,000 age-2, and 429,000 age-3/4 kokanee based on trawling (Table 36). The highest kokanee fry densities were observed in the northern portion of the lake (Section 1; Figure 36), particularly near Wolf Lodge Bay. We observed much lower abundance of fry in sections 2 and 3. The highest adult abundance was observed in Section 2. Kokanee sampled by trawling varied in length from 21–285 mm TL (Figure 38) and varied in age from 0–4 years old (Figure 39).

Lake Coeur d'Alene Spawner Assessment

Spawning kokanee varied in length from 275–360 mm TL and all were estimated to be either three or four years old. Similar to past years, female kokanee represented a smaller proportion of the sample (Figure 40). Mean TL was 315 mm (SD = 12.2) and 282 mm (SD = 7.0) for male and female kokanee, respectively. Overall mean TL was 312 mm (SD = 15.1). Mean TL of kokanee spawners in 2018 was higher than in 2017, and all sampled fish met or exceeded the adult length objective (Figure 41).

Spirit Lake Population Monitoring

We sampled a total of 166 kokanee by trawling in Spirit Lake. We estimated a total abundance of 311,506 kokanee. We estimated abundances of 172,543 age-0, 64,137age-1, 10,816 age-2, and 64,010 age-3 kokanee based on trawling (Table 37). We estimated a total density of around 537 kokanee/ha and a density of 110 age-3 kokanee/ha (Table 37). An average number of fry were sampled, and there did not appear to be any pattern in age-specific abundance around the lake; kokanee tended to be well-distributed across all transects. The weak 2016 and 2017 year-classes were confirmed by low abundance of age-1 and 2 fish. Kokanee sampled during trawling varied in length from 34–268 mm TL (Figure 42; mean = 143 [SD = 87.7]) and varied in age from 0–3 years old (Figure 43).

DISCUSSION

Lake Coeur d'Alene

The kokanee population in Lake Coeur d'Alene has supported a productive harvest fishery over the past several years, and angling was reportedly good again during 2017. In the past, the population has been negatively affected by adverse environmental conditions, namely spring flooding (Ryan et al. 2014); however, stable conditions in recent history have improved the population. Abundance of young-of-year kokanee, as indexed by trawling, appears to be lower than the 10-year mean, but more than 3-fold higher than in 2016. This pattern is consistent with age-0 abundance in Spirit Lake and could be a product of regional environmental conditions. Regardless of the cause, we expect that relatively weak year-classes produced during 2015–2016 will actually benefit the fishery by improving growth, and as a result, length-at-age of adults.

We found that adult spawner size exceeded the desired range and was above the most recent 20 year average (Figure 41). Our mean length estimate in 2018 (TL = 312 mm) was above the desired range and most adult kokanee were likely of desirable size to anglers. While potential management options for influencing the kokanee fishery are limited, continued population monitoring is important for understanding kokanee ecology and for providing public information.

Spirit Lake

Spirit Lake has historically been one of Idaho's top kokanee fishing waters (Ryan et al. 2014). The lake supports a summer troll fishery and winter ice fishery, making it an important regional resource. The kokanee population has a long history of being highly variable in terms of recruitment and growth, and this has continued over the last 15 years (Ryan et al. 2014). The fishery has tended to follow suit whereby angling effort tracks adult abundance and size structure; however, the fishery can be variable due to winter ice conditions as well. The variability in the fishery seems to have persisted in recent history. Spirit Lake does not have any pelagic predators, unlike other large northern Idaho lakes (i.e., Lake Pend Orielle, Lake Coeur d'Alene), so its kokanee population serves as a baseline for which other populations can be compared (Ryan et al. 2014). The absence of predators also allows kokanee to reach high densities in Spirit Lake. As such, the kokanee population often exhibits strong density-dependent growth, thus depressing size structure and leading to decreased interest among anglers.

Based on sampling in 2018, overall kokanee abundance has declined substantially compared to our most recent surveys. This pattern has likely been influenced by relatively poor recruitment during 2015–2016 and apparently high mortality of adults from age-2 to age-3 during 2016–2018. Prior to this time, high recruitment had created strong density-dependent growth and dramatically reduced size structure of the adult population. It has been demonstrated in other nearby systems (e.g., Dworshak Reservoir) that adult mortality can be high when density compromises body condition (Wilson et al. 2010). More age-3 kokanee are now surpassing 200 mm TL and mean length of age-3 fish was 243 mm. The relatively small size of adults has reduced angler interest largely because catchability can decrease in conjunction with adult length. Consistent with results from Lake Coeur d'Alene, we found that 2016 produced another weak year-class of kokanee in Spirit Lake. At this stage, several weak year-classes during 2015–2016 may benefit the fishery as long as recent cohorts sustain spawning stocks sufficient for replacement. Follow-up sampling should be conducted to better understand long-term trends in kokanee population abundance and size structure.

RECOMMENDATIONS

1. Continue annual kokanee population monitoring on Lake Coeur d'Alene and Spirit Lake.

Table 36. Estimated abundance of kokanee made by midwater trawl in Lake Coeur d'Alene, Idaho, from 1979–2018.

Year	Age class				Total
	Age-0	Age-1	Age-2	Age-3/4	
2018	1,003,259	503,060	58,008	428,884	1,993,211
2017	2,114,549	53,927	4,437,410	899,195	7,505,082
2016	690,170	729,709	2,461,281	1,306,550	2,967,710
2015	349,683	3,664,419	5,307,640	135,809	9,457,551
2014	2,877,209	2,153,877	2,790,295	319,080	8,140,461
2013	1,349,000	3,663,000	1,319,000	373,000	6,704,000
2012	--	--	--	--	--
2011	3,049,000	1,186,000	1,503,000	767,000	6,505,000
2010	660,400	2,164,100	1,613,300	506,200	4,943,900
2009	731,600	1,611,800	2,087,400	333,600	4,764,400
2008	3,035,000	3,610,000	1,755,000	28,000	8,428,000
2007	3,603,000	2,367,000	136,000	34,000	6,140,000
2006	7,343,000	1,532,000	91,000	33,900	8,999,000
2005	--	--	--	--	--
2004	7,379,000	1,064,000	141,500	202,400	8,787,000
2003	3,300,000	971,000	501,400	182,300	4,955,000
2002	3,507,000	934,000	695,200	70,800	5,207,000
2001	7,098,700	929,900	193,100	25,300	8,247,000
2000	4,184,800	783,700	168,700	75,300	5,212,600
1999	4,091,500	973,700	269,800	55,100	5,390,100
1998	3,625,000	355,000	87,000	78,000	4,145,000
1997	3,001,100	342,500	97,000	242,300	3,682,000
1996	4,019,600	30,300	342,400	1,414,100	5,806,400
1995	2,000,000	620,000	2,900,000	2,850,000	8,370,000

Table 36 (continued)

Year	Age 0	Age 1	Age 2	Age 3/4	Total
1994	5,950,000	5,400,000	4,900,000	500,000	12,600,000
1993	5,570,000	5,230,000	1,420,000	480,000	12,700,000
1992	3,020,000	810,000	510,000	980,000	5,320,000
1991	4,860,000	540,000	1,820,000	1,280,000	8,500,000
1990	3,000,000	590,000	2,480,000	1,320,000	7,390,000
1989	3,040,000	750,000	3,950,000	940,000	8,680,000
1988	3,420,000	3,060,000	2,810,000	610,000	10,900,000
1987	6,880,000	2,380,000	2,920,000	890,000	13,070,000
1986	2,170,000	2,590,000	1,830,000	720,000	7,310,000
1985	4,130,000	860,000	1,860,000	2,530,000	9,370,000
1984	700,000	1,170,000	1,890,000	800,000	4,560,000
1983	1,510,000	1,910,000	2,250,000	810,000	6,480,000
1982	4,530,000	2,360,000	1,380,000	930,000	9,200,000
1981	2,430,000	1,750,000	1,710,000	1,060,000	6,940,000
1980	1,860,000	1,680,000	1,950,000	1,060,000	6,500,000
1979	1,500,000	2,290,000	1,790,000	450,000	6,040,000

Table 37. Estimated abundance of kokanee made by midwater trawl in Spirit Lake, Idaho, from 1981–2018.

Year	Age class: Age-0	Age-1	Age-2	Age-3	Total	Age-3/ha
2018	172,543	64,137	10,816	64,010	311,506	133
2017	287,804	1,755	62,891	42,317	396,209	73
2016	11,940	28,332	307,544	30,612	378,428	53
2015	7,598	60,828	2,104,886	368,167	2,541,479	629
2014	44,295	720,648	653,945	231,356	1,650,245	396
2013	--	--	--	--	--	--
2012	--	--	--	--	--	--
2011	1,092,000	185,700	382,300	65,500	1,725,400	112
2010	138,200	459,900	88,800	61,600	748,500	105
2009	260,700	182,600	75,900	30,000	549,200	51
2008	281,600	274,400	188,800	56,400	801,200	96
2007	439,919	210,122	41,460	20,409	711,910	35
2006	--	--	--	--	--	--
2005	508,000	202,000	185,000	94,000	989,100	161
2001–04	--	--	--	-	--	--
2000	800,000	73,000	6,800	7,800	901,900	13
1999	286,900	9,700	50,400	34,800	381,800	61
1998	28,100	62,400	86,900	27,800	205,200	49
1997	187,300	132,200	65,600	6,500	391,600	11
1996	--	--	--	--	--	--
1995	39,800	129,400	30,500	81,400	281,100	142
1994	11,800	76,300	81,700	19,600	189,400	34
1993	52,400	244,100	114,400	11,500	422,400	20
1992	--	--	--	--	--	--
1991	458,400	215,600	90,000	26,000	790,000	45
1990	110,000	285,800	84,100	62,000	541,800	108
1989	111,900	116,400	196,000	86,000	510,400	150
1988	63,800	207,700	78,500	148,800	498,800	260
1987	42,800	164,800	332,800	71,700	612,100	125
1986	15,400	138,000	116,800	35,400	305,600	62
1985	149,600	184,900	101,000	66,600	502,100	116
1984	3,300	16,400	148,800	96,500	264,900	168
1983	111,200	224,000	111,200	39,200	485,700	68
1982	526,000	209,000	57,700	48,000	840,700	84
1981	281,300	73,400	82,100	92,600	529,400	162

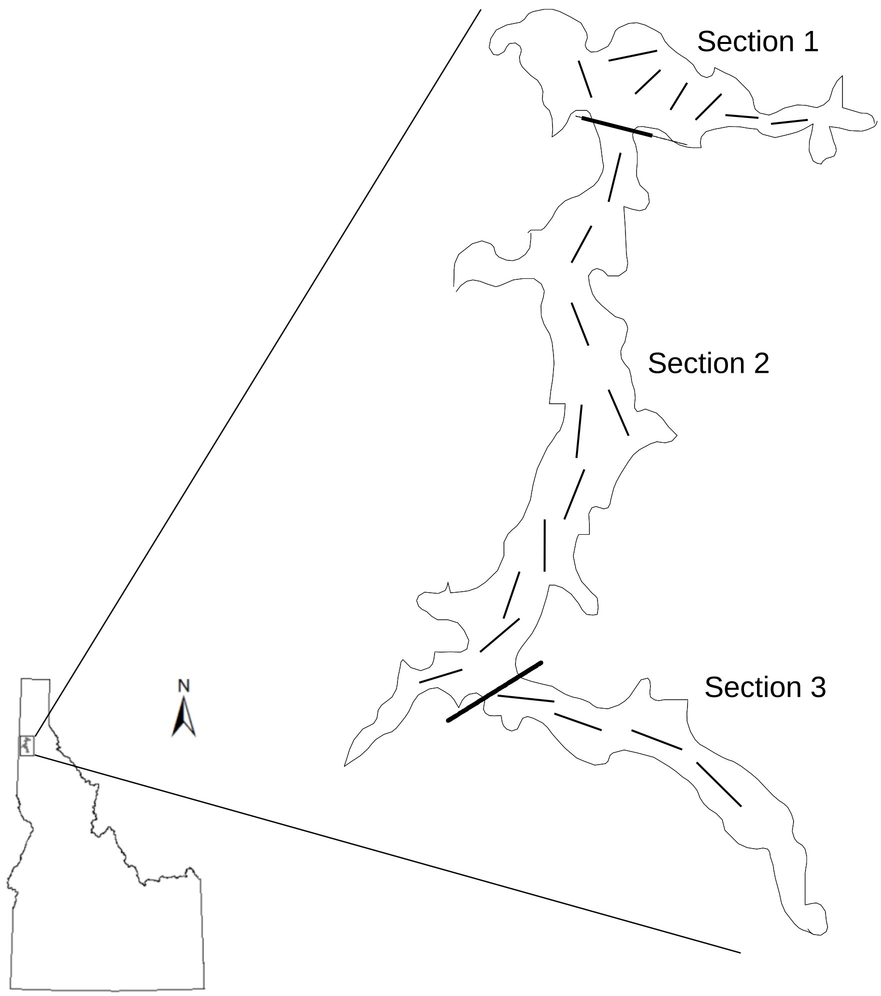

Figure 36. Approximate location of historical trawling transects used to estimate abundance of kokanee in Lake Coeur d’Alene, Idaho.

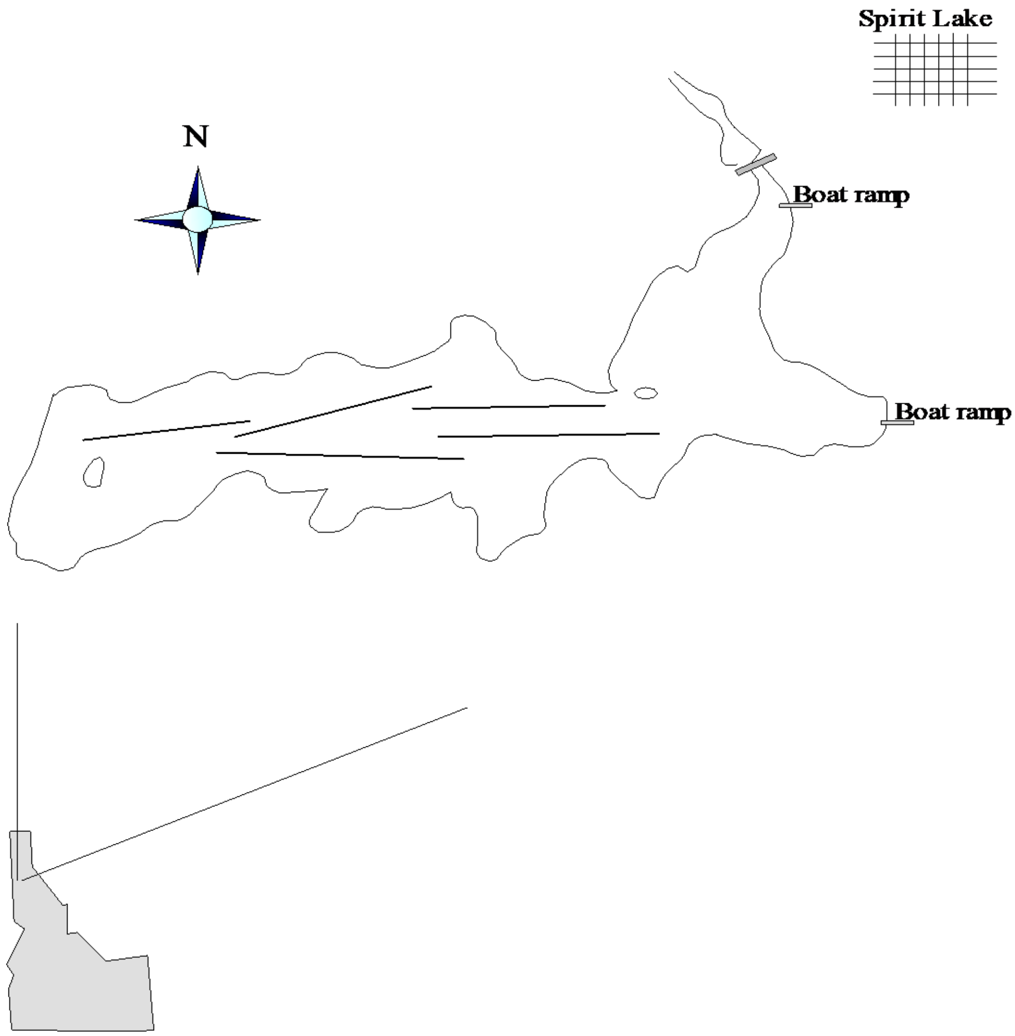

Figure 37. Approximate location of historical trawling transects used to estimate abundance of kokanee in Spirit Lake, Idaho.

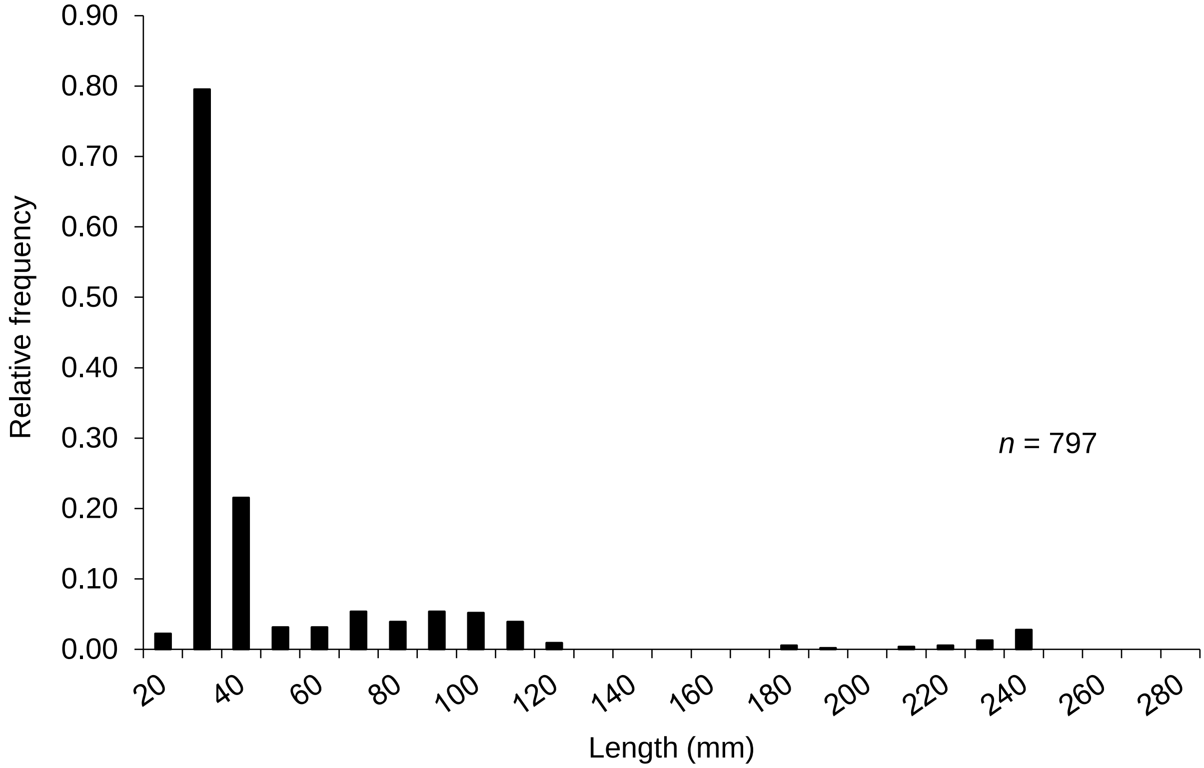

Figure 38. Length-frequency distribution for kokanee sampled using a modified-midwater trawl from Lake Coeur d’Alene, Idaho (July 6–7, 2018).

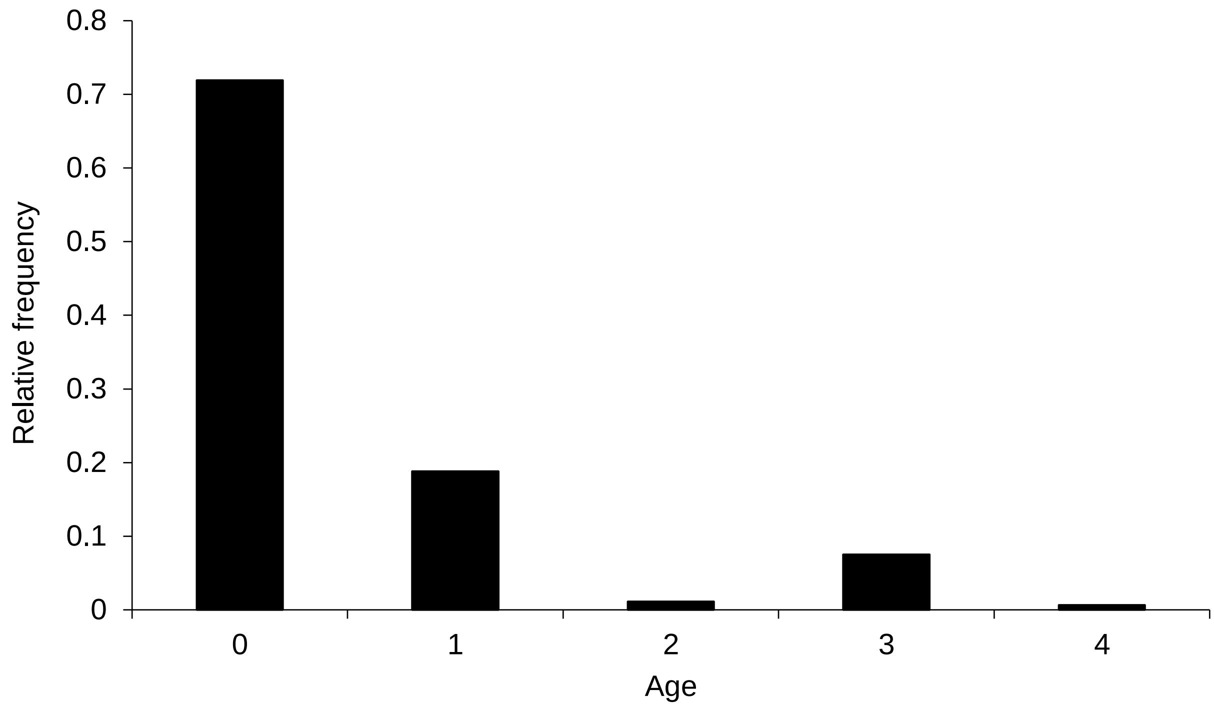

Figure 39. Age-frequency distribution for kokanee sampled using a modified-midwater trawl from Lake Coeur d’Alene, Idaho (July 6–7, 2018).

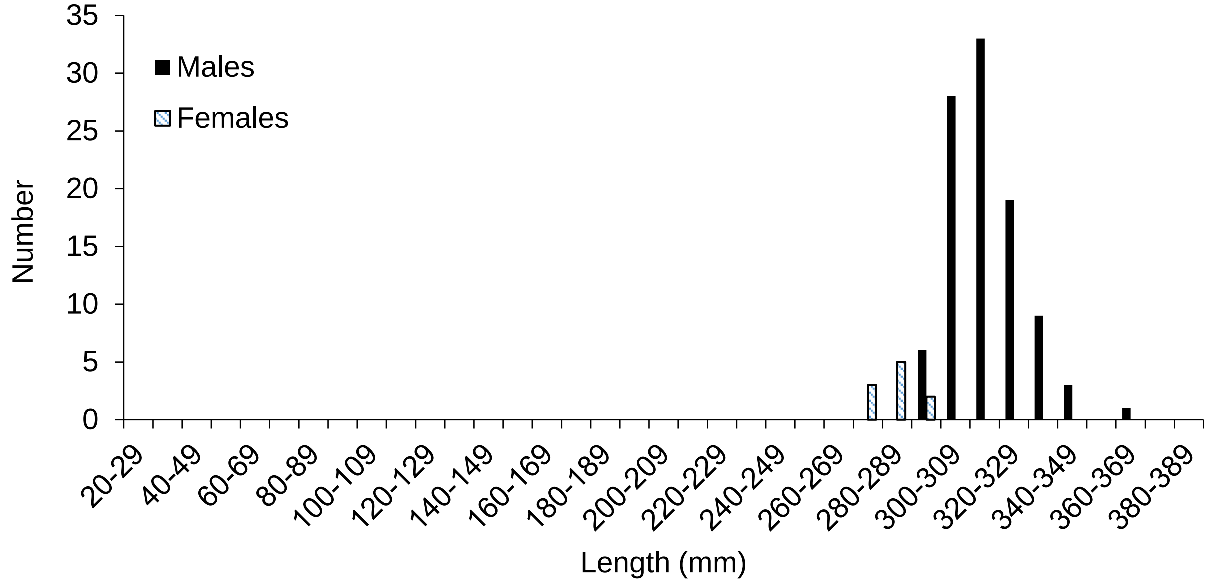

Figure 40. Length-frequency distribution for male and female kokanee sampled from Lake Coeur d'Alene, Idaho (December 7, 2018).

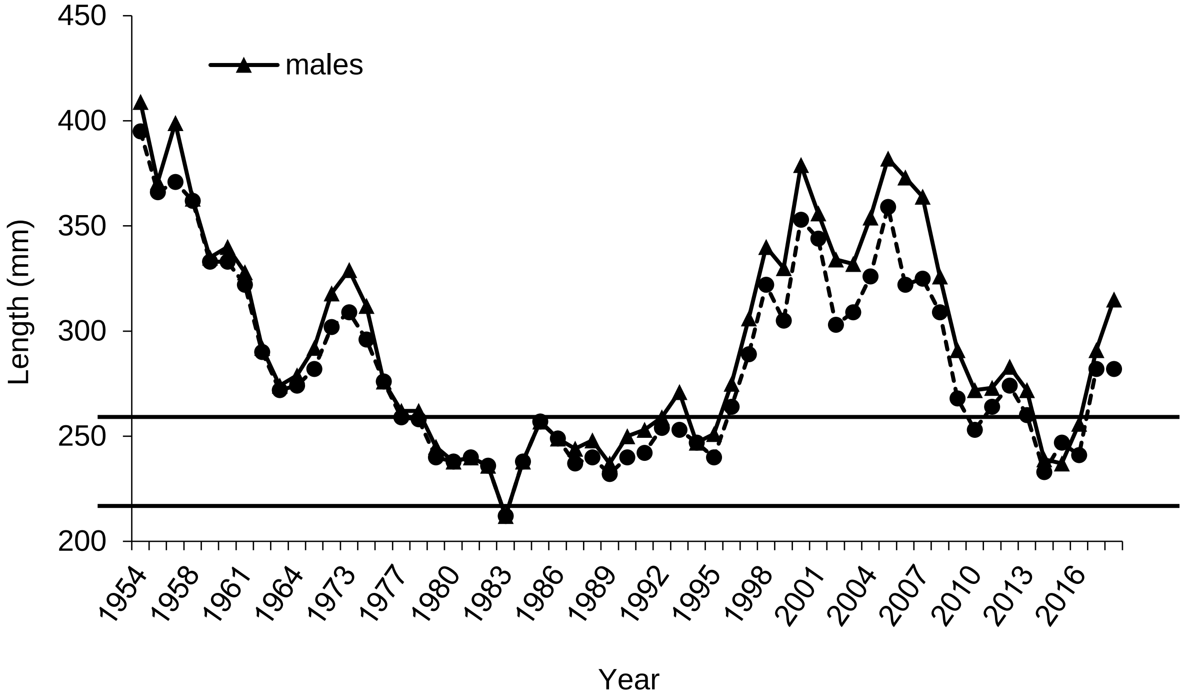

Figure 41. Mean total length of mature male and female kokanee sampled near Higgens Point in Lake Coeur d'Alene, Idaho (1954–2018). Horizontal lines indicate the upper and lower limit of the adult length management objective (250–280 mm).

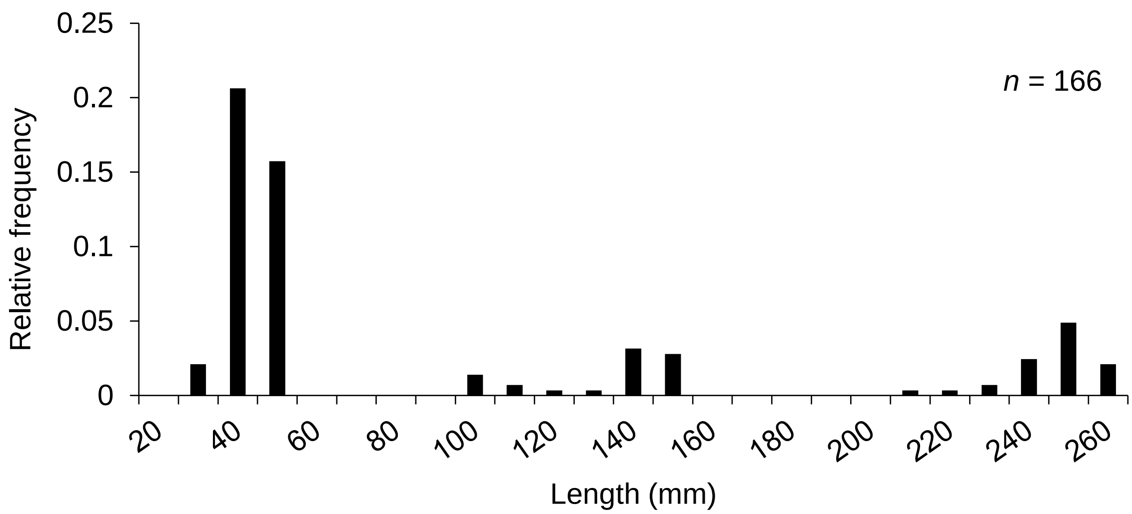

Figure 42. Length-frequency distribution for kokanee sampled using a modified-midwater trawl from Spirit Lake, Idaho (July 24, 2017).

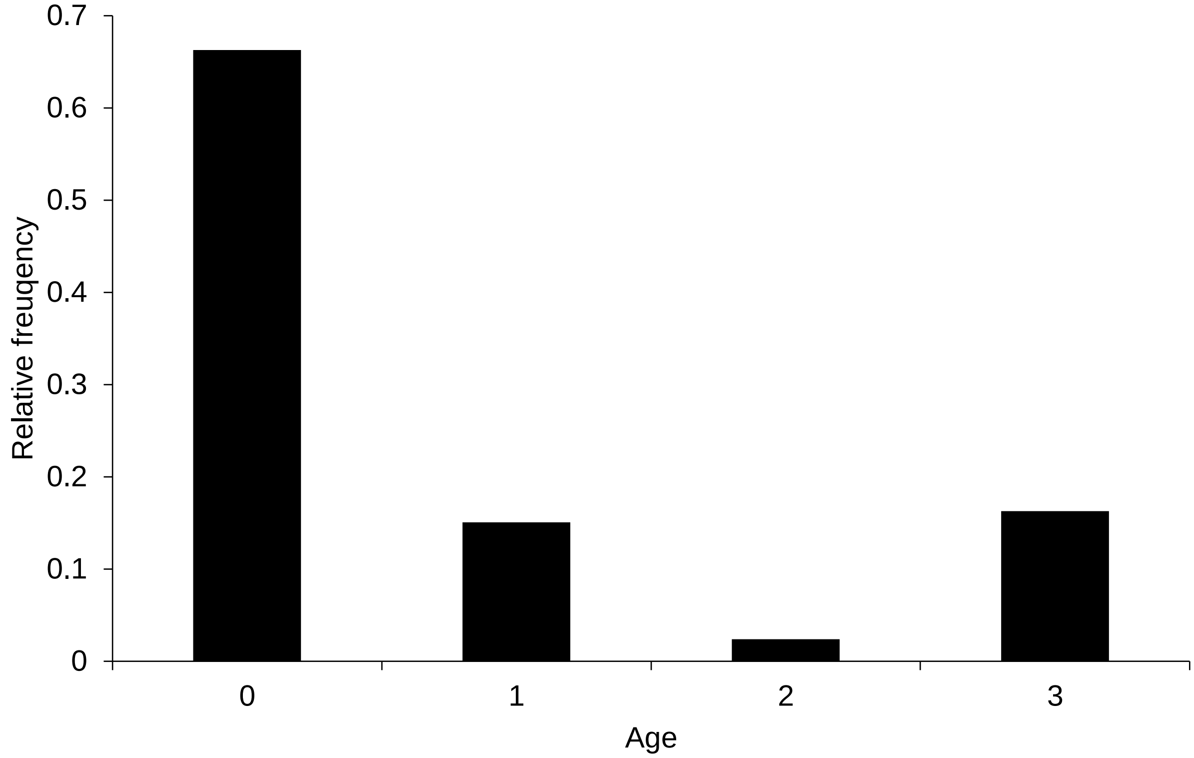

Figure 43. Age-frequency distribution for kokanee sampled using a modified-midwater trawl from Spirit Lake, Idaho (July 24, 2017).

LAKE COEUR D'ALENE CHINOOK SALMON EVALUATIONS

ABSTRACT

We evaluated escapement of Fall Chinook Salmon *Oncorhynchus tshawytscha* to index trends in adult abundance by enumerating redds at standard index reaches of the Coeur d'Alene and St. Joe rivers. In 2018, we observed a total of 28 redds at all index reaches combined. All redds were observed in the Coeur d'Alene River and none were observed in the St. Joe River. Redd abundance decreased substantially from a record high in 2015 across all index reaches. Chinook Salmon support an important recreational fishery in Lake Coeur d'Alene and also have strong potential to alter the pelagic prey (i.e., kokanee *O. nerka*) community, necessitating continued monitoring to understand changes to the fishery at-large. Future assessments should include annual monitoring of adult escapement and spawner age structure so that changes in abundance and age-at-maturity can be identified. Information related to population characteristics can be used to assess population-level changes and facilitate better management of the Lake Coeur d'Alene fishery.

In addition to adult abundance monitoring, we continued efforts to improve hatchery Fall Chinook Salmon performance. Similar to the previous four years, experimental fall outplants occurred during 2018 in Wolf Lodge Creek to improve relative return-to-creel. Stocking performance is anticipated to be evaluated using fishery-dependent data from angler logs kept by avid Chinook Salmon anglers. We recommend continued monitoring of hatchery fish performance using fishery-dependent data obtained from angler records. Additionally, improving performance of hatchery products and dispersing the fall fishery should remain a priority. Efforts to improve performance should focus on utilizing locally-adapted adlfuvial stocks to avoid post-smolting emigration. We recommend continuing efforts to develop locally-adapted broodstock sources to use for future supplementation of the fishery.

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INTRODUCTION

Chinook Salmon *Oncorhynchus tshawytscha* is an anadromous Pacific salmon species historically found in much of the Columbia River Basin (Wallace and Zaroban 2013). While anadromy is the natural life history form of Chinook Salmon, they have been successfully stocked into lentic systems outside of their native distribution where they exhibit adfuvial life histories. For example, both Chinook Salmon and Coho Salmon *O. kisutch* have been stocked into large lakes and reservoirs in the northern United States where they have naturalized and provide important angling opportunities (Diefenbach and Claramunt 2013; MFWP 2013). With adequate fluvial spawning habitat, many landlocked Pacific salmon populations are able to adopt adfluvial life history strategies and naturalize in lentic systems, persisting well outside of their native distribution.

Fall Chinook Salmon were first stocked into Lake Coeur d'Alene in 1982 as a biomanipulation tool to reduce kokanee *O. nerka* abundance. Kokanee exhibit density-dependent growth, and increases in population abundance commonly reduce length-at-age. This relationship has been evident in Lake Coeur d'Alene; fishery managers noted declines in size structure of kokanee during the late-1970s and concluded that fishing mortality could not sufficiently influence abundance. Goodnight and Mauser (1980) recommended an increase in the daily bag limit of kokanee from 25 to 50 fish following the 1979 season. The following year, Mauser and Horner (1982) noted that "the population size still exceeded the capacity of the system to produce fish of a desirable size to anglers" and recommended that predators be used to reduce abundance. Although kokanee harvest had reached an all-time high of ~578,000 fish in 1979, managers were convinced that improvements in size structure were needed to maintain angler interest. The semelparous life history and short life span of Chinook Salmon made it a desirable predator, and it was thought that their abundance could be regulated by stocking alone. An added benefit of Chinook Salmon was the creation of an additional fishery in the system. Previous managers had no expectation of naturalization and wild reproduction from Chinook Salmon introduced into Lake Coeur d'Alene; however, Chinook Salmon were observed spawning in Wolf Lodge Creek as early as 1984 and wild fish had become common in the fishery by 1986. Wild Chinook Salmon redds were observed in the Coeur d'Alene River and St. Joe River around 1988, and by then wild fish dominated the angler catch (Horner et al. 1989; Fredericks and Horner 1999).

The Idaho Department of Fish and Game (IDFG) continues to use Chinook Salmon as one tool for managing the kokanee population in Lake Coeur d'Alene. In addition, stocking supplements the fishery by providing additional harvest opportunity. The IDFG's management objective regarding Lake Coeur d'Alene has been to maintain predator stocking at a rate that does not depress the kokanee population, yet helps to achieve kokanee size structure objectives. Combinations of redd excavation and stocking (or lack thereof) have been used to regulate abundance for Chinook Salmon. Estimates of wild production have been obtained by coupling redd survey information with known egg-fry survival rates; subsequently, redds have been destroyed during some years to bring estimated production in line with objectives. Historically, Chinook Salmon redd objectives have been 100 total redds among both the Coeur d'Alene and St. Joe Rivers. During years when the objective was exceeded, redds have been excavated, and supplemental stocking has been used during years when wild redd abundance was below objective. However, the effectiveness of managing adult Chinook Salmon densities using supplemental stocking and redd excavation has been unsubstantiated. In addition, the kokanee population appears to be influenced more by environmental conditions rather than predator abundance. As such, in recent years the IDFG has not excavated Chinook Salmon redds, but monitors trends in redd abundance and supplemental stocking has been maintained at ~20,000 individuals annually since 2010 to supplement harvest.

One factor that has influenced the IDFG's ability to control adult Chinook Salmon abundance in Lake Coeur d'Alene is related to performance and retention of hatchery fish. Although 20,000 individuals are stocked annually, return-to-creel of hatchery fish is very low. Creel surveys conducted at angling tournaments and anecdotal evidence from avid Chinook Salmon anglers suggest that recruitment of hatchery fish to the fishery is close to zero. Maiolie and Fredericks (2014) evaluated performance of hatchery Chinook Salmon among rearing hatcheries and between spring and fall stocking seasons. The authors reported that hatchery fish performance may be lower among cohorts that were raised at Nampa Fish Hatchery and released in spring stocking groups. These results have influenced current management, and the IDFG now rears supplemental Chinook Salmon for Lake Coeur d'Alene at Cabinet Gorge Hatchery in Clark Fork, Idaho. In addition, stocking has been moved to early fall (i.e., late-September or early-October) when fish are larger and near smoltification. Anglers have reported that hatchery Chinook Salmon (identified by a clipped adipose fin) were more commonly encountered during 2013–2014, suggesting that those individuals are now recruiting to the fishery at higher rates, but perhaps still at lower rates than desired by managers.

Because Chinook Salmon occur naturally with anadromous life histories, it is likely that many are entrained shortly after release. Pacific Salmon demonstrate strong homing behavior and life history fidelity. However, bypassing critical early life stages (i.e., smoltification), imprinting of juveniles, or stocking brood derived from locally-adapted individuals may be used to overcome this tendency. By stocking after smolting occurs and simulating migration from a lotic to lentic environment, managers may be able to impose an adfluvial life history on hatchery stock. Mimicking a migratory life history and imprinting juveniles to a fluvial, "natal" environment is critical for altering the life history of anadromous fishes. For example, Alaska Department of Fish and Game (ADFG) has documented low retention of anadromous fishes stocked directly into freshwater lakes. In contrast, ADFG has obtained higher retention and higher return-to-creel among groups that are held in lake tributaries, imprinted, and allowed to emigrate to the respective lake where they carry out their adult life history (Havens et al. 1987). An additional hypothesis is that smolt-related emigration can be reduced by using locally-adapted adfluvial broodstock. The utilization of locally-adapted brood has been demonstrated in many systems, especially in anadromous fish populations (Taniguchi 2003), and may likely increase retention of hatchery Chinook Salmon in Lake Coeur d'Alene.

Both kokanee and Chinook Salmon provide popular angling opportunities in Lake Coeur d'Alene. The IDFG's objective for Lake Coeur d'Alene is to manage for a kokanee yield fishery (15 fish daily bag limit) and trophy Chinook Salmon fishery (2 fish daily bag; none under 508 mm). Prior to the introduction of Chinook Salmon, nearly all (~99%) of the angling effort in Lake Coeur d'Alene has been targeted at kokanee (Rieman and LaBolle 1980); however, more recent studies have shown that most effort (~42%) is now targeting Chinook Salmon (Hardy et al. 2010). Chinook Salmon are highly-desired by anglers because they often grow to trophy sizes and have very palatable flesh. As such, monitoring the Chinook Salmon population and understanding factors that regulate it is critical for providing quality angling opportunities.

OBJECTIVES

1. Monitor trends in Chinook Salmon redd abundance as an index to adult abundance.

2. Evaluate stocks and stocking strategies for hatchery Chinook Salmon to improve return-to-creel of supplemental fish.

STUDY AREA

Lake Coeur d'Alene is a natural mesotrophic water body located in the Panhandle of northern Idaho (Figure 44). Lake Coeur d'Alene lies within Kootenai and Benewah counties and it is the second largest natural lake in Idaho with a surface area of 12,742 ha, mean depth of 24 m, and maximum depth of 61 m (Rich 1992). The Coeur d'Alene and St. Joe rivers are the major tributaries to Lake Coeur d'Alene; however, many smaller second and third order tributaries also exist. The outlet to Lake Coeur d'Alene is the Spokane River, a major tributary to the Columbia River. Water resource development in the watershed includes Post Falls Dam, which was constructed on the Spokane River in 1906, and raised the lake level approximately 2.5 m.

The fish assemblage in Lake Coeur d'Alene is composed of three native sport fish species, five native nongame species, 16 introduced sport fish species, and one introduced nongame species. The fishery in the lake, however, can be broadly summarized as belonging to one of three components—kokanee, Chinook Salmon, or littoral species; all of these components are popular among anglers. Increased fish assemblage complexity has undoubtedly resulted in increased biological interactions, but also diversified angler opportunity. Because of its close proximity to several major cities (i.e., Coeur d'Alene; Spokane), Lake Coeur d'Alene generates high angling effort, contributing considerably to both state and local economies.

METHODS

Spawner Abundance

Chinook Salmon escapement has been monitored using annual redd counts in the Coeur d'Alene and St. Joe rivers since 1990. Standardized index reaches (Table 38) have been sampled annually sometime during late September–early October to estimate relative redd abundance. Early surveys were done via helicopter, but since 2012 surveys have been conducted by watercraft (Maiolie and Fredericks 2014). Two individuals floated the Coeur d'Alene River index reaches during October 5, 2018 and the St. Joe index reach during October 8, 2018 using a drift boat. During sampling, all redds were enumerated and georeferenced with a global positioning system. Redd abundance was estimated as the total number of redds observed among all index reaches. We compared among previous years' surveys to provide insight on trends in abundance.

Performance of Supplemental Chinook Salmon

Eggs from Tule Fall Chinook Salmon were obtained from Fort Peck State Fish Hatchery located near Fort Peck, Montana, and were hatched and reared at Cabinet Gorge Hatchery in Clark Fork, Idaho. The adipose fin was completely removed from all individuals (n = 24,460), but they were not marked as in previous years. Hatchery individuals were stocked into Wolf Lodge Creek (Figure 44) on September 17–18, 2018. Hatchery Chinook Salmon were stocked post-smoltification and in an upstream location along Wolf Lodge Creek to improve homing behavior and survival. All individuals were thermal marked by Cabinet Gorge Fish Hatchery staff; marks may be used to assign sampled adults back to brood year and to differentiate among stocking strategies.

RESULTS

We summarized redd abundance to monitor adult escapement and trends in natural production. We observed a total of 105 redds at index reaches in the Coeur d'Alene River basin. Of these, we observed 27 redds in the mainstem Coeur d'Alene River between Cataldo and the confluence of the South Fork Coeur d'Alene River, and 1 redd in the North Fork Coeur d'Alene River between the confluence of the South Fork Coeur d'Alene River and the confluence of the Little North Fork Coeur d'Alene River (Table 38). We did not sample the South Fork Coeur d'Alene River during 2018 due to logistical constraints associated with low flow conditions. No redds were observed in the St. Joe River between St. Joe City and the Calder Bridge (Table 38). Chinook Salmon redd abundance has decreased around 91% since high abundance observed in 2015 (Figure 45).

DISCUSSION

The wild Chinook Salmon fishery has increased in abundance over the past two decades, although 2018 produced poor angling by anecdotal assessment. The combination of several factors (i.e., stable environmental conditions, abundant forage [kokanee]) has likely allowed the population to rebound from the low abundances observed in the late-1990s (Watkins et al. *in review*). However, the most recent redd survey (fall 2018) showed that adult escapement was well below the long-term average (82 redds) and in consistent decline since 2015.

The Chinook Salmon fishery in Lake Coeur d'Alene has historically been supported almost entirely by naturally-produced individuals. Anecdotal evidence from anglers suggests that age-1 and age-2 adipose-clipped individuals have been more common in the fishery in recent history. The IDFG has made the following advances in Chinook Salmon rearing and stocking which may be contributing to improved performance of hatchery individuals: 1) Fall Chinook Salmon rearing has been moved from Nampa Hatchery to Cabinet Gorge Hatchery where rearing temperatures are colder and the transport distance to Lake Coeur d'Alene is shorter, and 2) size-at-release has been improved by switching from spring to fall stocking. The combination of changes in rearing and release timing are expected to improve survival of hatchery fish; however, we will be unable to fully-quantify the effect of these management actions until 2014 outplants recruit to the fishery. While the direct results of these actions are difficult to substantiate, we cannot attribute this change in occurrence of hatchery individuals to any other major management changes. This is consistent with previous studies showing that performance of hatchery fish is often directly related to length-at-release where larger individuals typically exhibit higher survival and return-to-creel than their smaller counterparts (Henderson and Cass 2011).

Despite ongoing efforts to identify factors influencing return-to-creel of hatchery produced Chinook Salmon, the post-release fate of those individuals remains unknown. Previous research has addressed factors that limit survival (Maiolie and Fredericks 2013; Maiolie and Fredericks 2014), but no work has sought to understand retention of hatchery-origin Chinook Salmon and whether post-release emigration may be a limiting factor. Future work will be aimed at evaluating relative return-to-creel by comparing stocking strategies that are hypothesized to improve retention. Anglers often catch adipose-removed Chinook Salmon in Lake Roosevelt which have presumably emigrated from Lake Coeur d'Alene and become entrained in that reservoir (William Baker, Washington Department of Fish and Wildlife, personal communication). These reports are not uncommon and are received from both anglers and Washington Department of Fish and Wildlife personnel. Post-release emigration has been documented in other lentic systems in Idaho where Fall Chinook Salmon are stocked. For instance, hatchery Chinook Salmon stocked into

Deadwood Reservoir in the Southwest Region have been sampled in Black Canyon Reservoir on the Payette River (Koenig et al. 2015). Additionally, hatchery Chinook Salmon stocked into Anderson Ranch Reservoir have been reported in Arrowrock Reservoir and Lucky Peak Reservoir (Arthur Butts, personal communication). This raises serious concern about post-release retention of hatchery stock and its effect on return-to-creel. It is likely that Chinook Salmon from anadromous stocks have a strong tendency to emigrate after release, particularly when stocked into waters within the Columbia River Basin. The maintenance of this life history may lead to a substantial portion of the hatchery fish attempting to emigrate after release. Improving retention will likely require the use of a method that imposes an adfluvial life history on hatchery individuals, or require the use of a landlocked, adfluvial stock (i.e., Lake Coeur d'Alene) for hatchery production. As such, beginning in 2018, IDFG began stocking landlocked Fall Chinook from Fort Peck Fish Hatchery, MT. Efforts to obtain eyed Fall Chinook eggs from Montana Fish, Wildlife, and Parks will continue and stocking efforts will focus on developing local, adfluvial broodstock sources for future supplementation.

RECOMMENDATIONS

1. Continue evaluation of hatchery Chinook Salmon performance; specifically, the influence of alternative stocks and stocking strategies.

2. Continue to enumerate Chinook Salmon redds at index reaches in the Coeur d'Alene River and St. Joe River.

Table 38. Location, description of index reaches, and number of Chinook Salmon redds counted during surveys from the most recent five years. Surveys are conducted in the Coeur d'Alene and St. Joe rivers. Only reaches with a long time series of information used to index Chinook Salmon redd abundance are included.

		Year				
Reach	Description	2018	2017	2016	2015	2014
	Coeur d'Alene River					
CDA 1	Cataldo to S.F. Coeur d'Alene River confluence	27	61	76	210	104
CDA 2	S.F. to L.N.F Coeur d'Alene River confluence	1	18	29	68	62
CDA 3	S.F. Coeur d'Alene River	--	--	--	10	4
	St. Joe River					
SJR 1	St. Joe City to Calder bridge	0	0	0	15	9

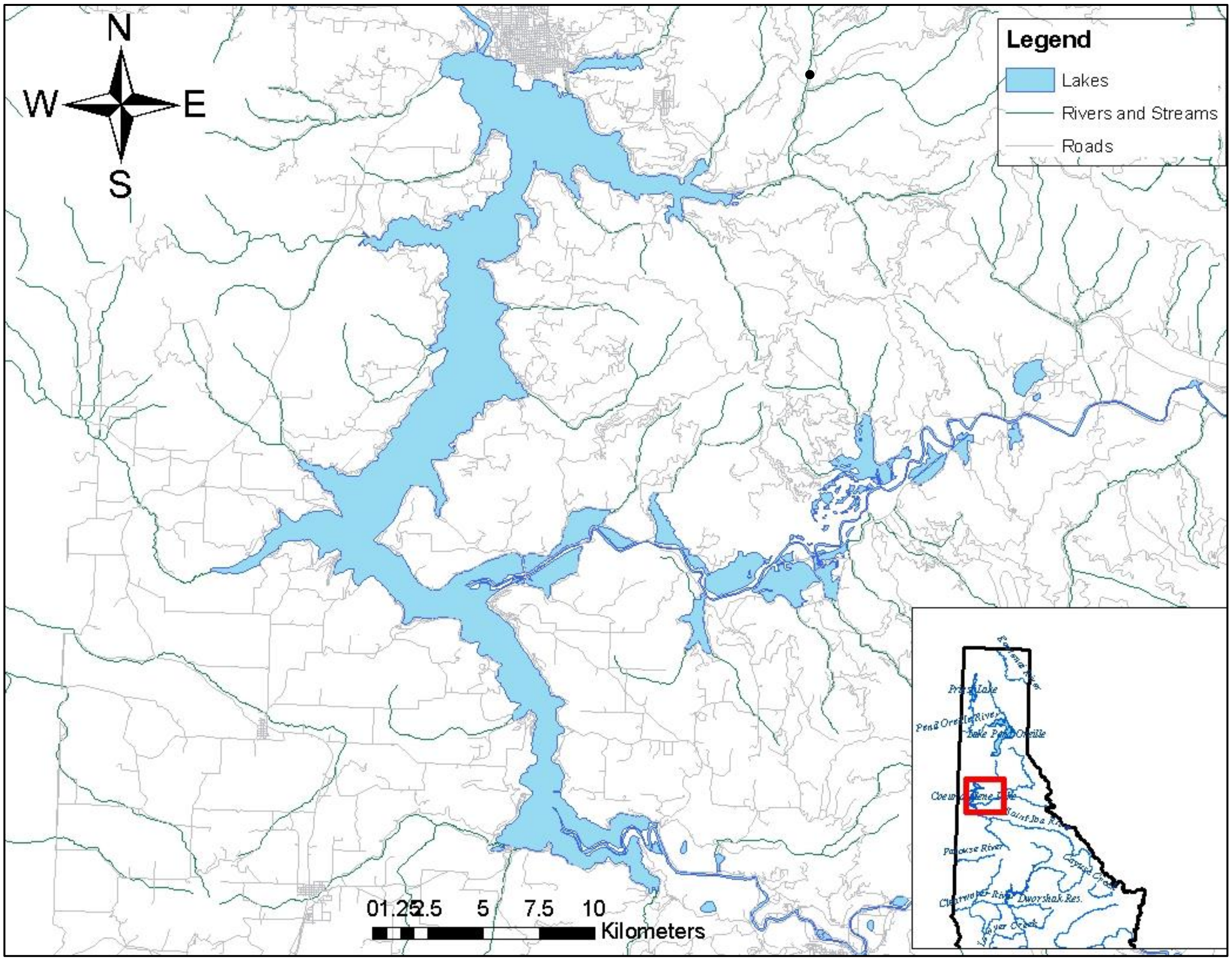

Figure 44. Location of Lake Coeur d'Alene, Idaho. The black dot on Wolf Lodge Creek represents the location of where juvenile hatchery Chinook Salmon were released.

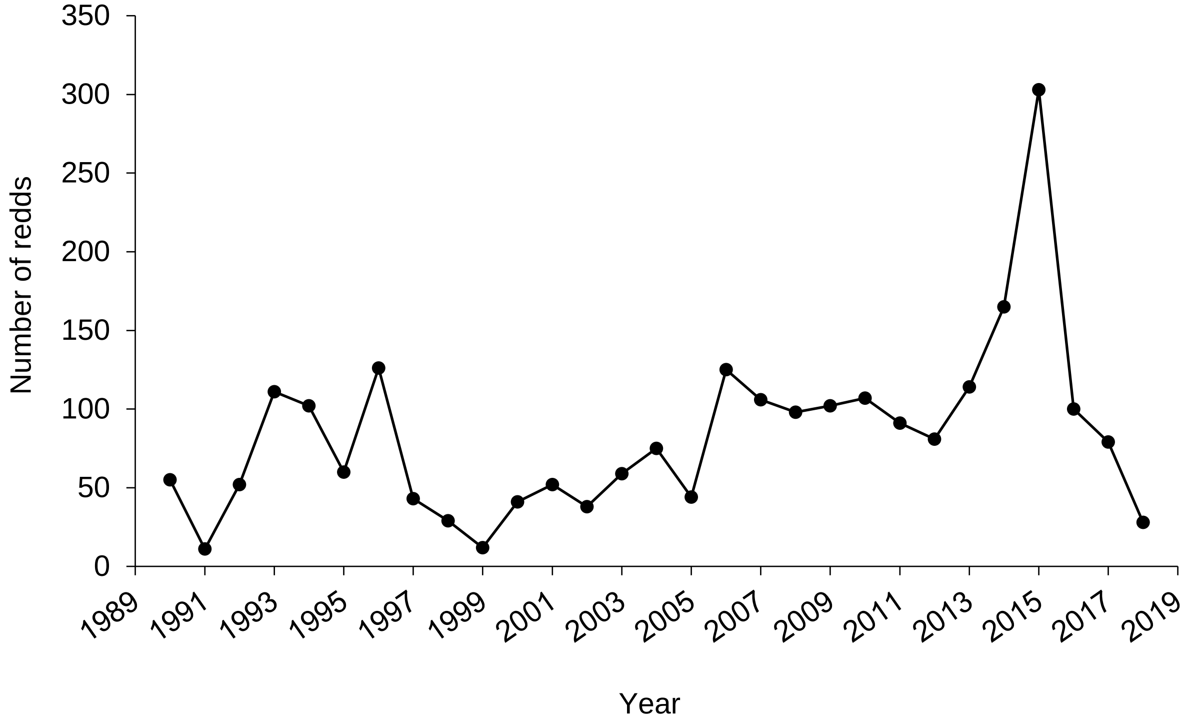

Figure 45. Number of Chinook Salmon redds counted during sampling of index reaches in the Coeur d'Alene River and St. Joe River from 1990–2018.

SPOKANE BASIN WILD TROUT MONITORING

ABSTRACT

Long-term data obtained from historical snorkeling transects have been critical for informing management of wild salmonids in the upper Spokane River Basin over the past several decades. In the Coeur d'Alene and St. Joe rivers, maintenance of long-term datasets has allowed the Idaho Department of Fish and Game to document responses of Westslope Cutthroat Trout *Oncorhynchus clarki lewisi* to environmental conditions, habitat rehabilitation, and angling regulations. During July 30–August 8, 2018, we used daytime snorkeling to observe fishes at historical sampling transects in the Coeur d'Alene River (n = 44) and St. Joe River (n = 35) basins. We estimated total Westslope Cutthroat Trout densities of 1.17 fish/100 m^2 in the North Fork Coeur d'Alene River (including Teepee Creek), 0.99 fish/100 m^2 in the Little North Fork Coeur d'Alene River, and 1.79 fish/100 m^2 in the St. Joe River. For Westslope Cutthroat Trout ≥ 300 mm in total length, we estimated densities of 0.31 fish/100 m^2 in the North Fork Coeur d'Alene River, 0.25 fish/100 m^2 in the Little North Fork Coeur d'Alene River, and 0.60 fish/100 m^2 in the St. Joe River. Densities of Rainbow Trout *O. mykiss* remained relatively low in both drainages, with estimates being similar to the past 15–20 years. Size structure of Westslope Cutthroat Trout continued to be slightly better in the St. Joe River compared to the Coeur d'Alene River system. Overall, trends in abundance and size structure of Westslope Cutthroat Trout in the upper Spokane River Basin have increased substantially over the past two decades and abundance continues to be variable, yet relatively high. Future monitoring should continue in order to better inform management of Westslope Cutthroat Trout and to demonstrate progress toward conservation objectives. Current catch-and-release angling regulations for Westslope Cutthroat Trout and liberal harvest regulations for non-native salmonids (i.e., Rainbow Trout, Brook Trout *Salvelinus fontinalis*) appear to be effective conservation measures for Westslope Cutthroat Trout.

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INTRODUCTION

Westslope Cutthroat Trout *Oncorhynchus clarki lewisi* is one of 14 subspecies of Cutthroat Trout *O. clarki* native to North America. The native distribution of Westslope Cutthroat Trout is the most widespread of the 14 subspecies spanning both sides of the Continental Divide (Behnke 1992; Behnke 2002). Their native distribution west of the Continental Divide includes the Salmon River and its tributaries, as well as all major drainages throughout the Idaho Panhandle. Despite their widespread distribution, declines in occurrence and abundance of Westslope Cutthroat Trout have been documented throughout their native range (Shepard et al. 2005). In Idaho, Westslope Cutthroat Trout still occupy 85% of their historical range (Wallace and Zaroban 2013). However, populations of Westslope Cutthroat Trout have been negatively influenced for a variety of reasons. Extensive land- and water-development activities, which have reduced available instream habitat and altered flows and thermal regimes, have negatively affected Westslope Cutthroat Trout (Peterson et al. 2010). Another important factor related to range and abundance reductions has been interaction with nonnative salmonids (i.e., Rainbow Trout *O. mykiss*, Brook Trout *Salvelinus fontinalis*), which often leads to competition and hybridization (Rainbow Trout only; Marnell 1988; Allendorf et al. 2004; Shepard et al. 2005; Muhlfeld et al. 2009).

Concerns about the rangewide status of Westslope Cutthroat Trout have resulted in two petitions for listing under the U.S. Endangered Species Act (ESA 1973, as amended) in 1997 and 2001. Subsequent evaluations of extant populations determined that the relatively broad distribution and persistence of isolated populations in Oregon, Washington, and Canada did not warrant protection under the ESA (U.S. Federal Register 1998, 2003). However, the U.S. Forest Service and Bureau of Land Management regard Westslope Cutthroat Trout as a sensitive species. Due to their importance as a recreational, cultural, and socioeconomic resource, the IDFG has intensely managed Westslope Cutthroat Trout populations for both general conservation and to provide quality angling opportunities.

The Spokane River Basin represents one of the most important areas for Westslope Cutthroat Trout conservation in Idaho and the Pacific Northwest; specifically, because major tributaries to the Spokane River (i.e., Coeur d'Alene River, St. Joe River) provide strongholds for this sensitive species (DuPont et al. 2009; Stevens and DuPont 2011). In addition, Westslope Cutthroat Trout populations in the upper Spokane River Basin support important recreational fisheries. The close proximity of the Coeur d'Alene and St. Joe rivers to large communities (i.e., Coeur d'Alene, Spokane) makes these waters popular destination trout fisheries, and angling pressure has increased in recent times (Fredericks et al. 1997; DuPont et al. 2009).

Over the past century, Westslope Cutthroat Trout angling regulations have become increasingly conservative with a shift toward catch-and-release angling (Hardy and Fredericks 2009; Kennedy and Meyer 2015). For example, prior to 2008, the lower portions of the Coeur d'Alene River (Lake Coeur d'Alene to confluence of Yellow Dog Creek) and St. Joe River (Lake Coeur d'Alene to confluence of North Fork St. Joe River) were managed under a two-trout daily bag and slot limit (none between 203–406 mm; Hardy and Fredericks 2009). However, currently the entire Spokane River Basin within Idaho is managed under a catch-and-release regulation for Westslope Cutthroat Trout, with the exception of the St. Maries River (two trout daily bag limit). The shift to catch-and-release rules led to improvements in these populations; however, increased education, enforcement of regulations, and habitat rehabilitation have also contributed. Westslope Cutthroat Trout populations responded positively to regulation changes and angler use increased. Improvements in the quality of the fishery, combined with the elimination of season restrictions, also increased angler use in the Coeur d'Alene and St. Joe rivers (IDFG 2013). Long-term monitoring has been tremendously important for formulating effective management plans for conservation of Westslope Cutthroat Trout in Idaho. Standardized monitoring has allowed IDFG to evaluate population-level responses to environmental change and management activities (Copeland and Meyer 2011; Kennedy and Meyer 2015), and thus improve the quality of the fishery in the Spokane River Basin.

OBJECTIVES

1. Monitor trends in abundance, distribution, and size structure of wild salmonids in the upper Spokane River Basin, with focus on Westslope Cutthroat Trout populations.

2. Monitor fish assemblage structure and species distribution to identify shifts that may occur for native and non-native fishes alike.

3. Maintain long-term trend data to provide information related to management of Westslope Cutthroat Trout.

STUDY AREA

The Coeur d'Alene and St. Joe rivers are the largest tributaries to Lake Coeur d'Alene and combined these drainages comprise ~50% of the greater Spokane River watershed. Both rivers originate in the Bitterroot Mountains along the Idaho-Montana border and are greatly influenced by spring runoff and snowmelt. Approximately 90% of the land area within the drainages is publically-owned and managed by the U.S. Forest Service (Strong and Webb 1970). Dominant land-use practices in both drainages include hard rock and placer mining and extensive timber harvest (Strong and Webb 1970; Quigley 1996; DEQ 2001). While the combination of these activities has negatively influenced instream habitat and water quality, increased oversight and regulation of land-use have improved environmental conditions for native fishes in both the Coeur d'Alene and St Joe river drainages (DEQ 2001).

Historical sampling reaches were established on the Coeur d'Alene River in 1973 (n = 42; Figure 46; Bowler 1974) and St Joe River in 1969 (n = 35; Figure 47; Rankel 1971; Davis et al. 1996). Sampling has been conducted on an annual basis for each reach since the beginning of the monitoring program, with the exception of seven reaches added to the St. Joe River in 1996 (Davis et al. 1996). Sampling reaches in the St. Joe River drainage occur only along the mainstem St. Joe River (Figure 47), while reaches within the Coeur d'Alene River drainage occur on the North Fork Coeur d'Alene River, Little North Fork Coeur d'Alene River, and Teepee Creek (Figure 46).

METHODS

Standard index reaches in the North Fork of the Coeur d'Alene (including Teepee Creek), Little North Fork Coeur d'Alene, and St. Joe rivers were sampled during July 30–August 8, 2018 using daytime snorkeling (DuPont et al. 2009; Thurow 1994). One (wetted width ≤10 m wide) or two (wetted width >10 m wide) observers slowly snorkeled downstream identifying fishes to species and estimating total length (TL; inches) of all salmonid species. All snorkelers obtained training on observation techniques and protocol by an experienced individual prior to conducting the survey. Transects have been permanently marked with a global positioning system (GPS) and digital photographs provided reference to the upper and lower terminus of each reach. Estimates of salmonid abundance was limited to age-1+ fish, as summer counts for young-of-year (YOY) Westslope Cutthroat Trout and Rainbow Trout are typically unreliable. After completion of each sampling reach, each species was enumerated and salmonid species (i.e., Westslope Cutthroat Trout, Rainbow Trout, Mountain Whitefish *Prosopium williamsoni*) were separated into 75 mm length groups. Nongame fish species (e.g., *Cottus* spp. and *Catostomus* spp.) were enumerated, but lengths were not estimated.

Reach length and wetted width were measured at each sampling site with a laser rangefinder. The habitat type (pool, riffle, run, glide, pocket water), maximum depth, dominant cover type and amount of cover (estimated as % of surface area) in the area sampled was measured to assess if changes in habitat were responsible for any changes in fish abundance and assemblage structure. Surface area (m^2) was estimated at each site to provide a measure of sampling effort. The number of salmonids observed was divided by the surface area sampled to provide a standardized relative abundance measure. We calculated a mean relative density that could be compared to previous years (DuPont et al. 2009). Non-target species were enumerated and reported as the total number observed.

Size structure of Westslope Cutthroat Trout was also estimated for each river system. Relative size distribution (RSD) was used to summarize length-frequency distributions (Neumann et al. 2012) and describe size structure. Relative size distribution was calculated as

$$\mathrm{RSD} = (a \,/\, b) \times 100,$$

where *a* is the number of fish greater than or equal to the minimum quality length and *b* is the number of fish greater than or equal to 305 mm length (Neumann and Allen 2007; Neumann et al. 2012).

RESULTS

North Fork Coeur d'Alene River

A total of 941 Westslope Cutthroat Trout, 31 Rainbow Trout, and 2,142 Mountain Whitefish was observed among the 44 sampling sites in the North Fork Coeur d'Alene River drainage. In addition, we observed 17 Northern Pikeminnow *Ptychocheilus oregonsis*, 12 Largescale Sucker *Catostomous macrocheilus*, and 10 Brook Trout. Mean total density of Westslope Cutthroat Trout was 1.17 fish/100 m^2 in the North Fork Coeur d'Alene River (including Teepee Creek) and 0.99 fish/100m^2 in the Little North Fork Coeur d'Alene River (Figure 48). Mean density of Westslope Cutthroat Trout ≥300 mm was 0.31 fish/100 m^2 in the North Fork Coeur d'Alene River and 0.25 fish/m^2 in the Little North Fork Coeur d'Alene River (Figure 49). For Westslope Cutthroat Trout during 2018, the mean estimates of total density and density of fish ≥300 mm were higher than the previous 10-year average (total Westslope Cutthroat Trout = 1.06 fish/100 m^2; Westslope Cutthroat Trout ≥ 300 mm = 0.24 fish/100 m^2) in the combined reaches. Mean total density of Rainbow Trout in the North Fork Coeur d'Alene River was 0.01 fish/100 m^2 and 0.27 fish/100m^2 in the Little North Fork Coeur d'Alene River (Figure 50). Mean total density of Mountain Whitefish was 2.06 fish/100 m^2 in the North Fork Coeur d'Alene River and 0.14 fish/100 m^2 in the Little North Fork Coeur d'Alene River (Figure 51). We estimated a RSD-305 of 41 for the Coeur d'Alene River Basin (Figure 56).

St. Joe River

A total of 790 Westslope Cutthroat Trout, zero Rainbow Trout, and 1,102 Mountain Whitefish was observed among the 35 sampling sites in the St. Joe River. In addition, we observed 249 Largescale Sucker, 264 Northern Pikeminnow, and five Bull Trout *S. confluentus* during 2018 sampling. Mean total density of Westslope Cutthroat Trout was 1.79 fish/100 m^2 (Figure 52). Mean density of Westslope Cutthroat Trout ≥300 mm was 0.60 fish/100 m^2 (Figure 53). The estimates of mean total Westslope Cutthroat trout density and density of fish ≥300 mm were slightly lower during 2018 than the previous 10-year averages of 1.81 fish/100 m^2 and 0.62 fish/100 m^2. Mean total density of Rainbow Trout and Mountain Whitefish was zero fish/100 m^2 and 1.75 fish/100 m^2, respectively (Figures 54 and 55). Size structure of Westslope Cutthroat Trout in the St. Joe River (RSD-305 = 56) was higher than in the Coeur d'Alene River Basin (Figure 56).

DISCUSSION

The upper Spokane River Basin represents one of Idaho's most important systems for conservation of Westslope Cutthroat Trout. Previous work on Westslope Cutthroat Trout showed that declines in abundance and size structure in both the Coeur d'Alene and St. Joe rivers were directly related to recruitment overfishing and habitat degradation (Rankel 1971; Mink et al. 1971; Lewynsky 1986). However, in the Spokane River Basin and elsewhere in Idaho, Westslope Cutthroat Trout populations have positively responded to changes in angling regulations and habitat quality.

Westslope Cutthroat Trout densities have increased markedly since the beginning of this monitoring program and continue to show improvement (Maiolie and Fredericks 2014). Although we have documented a considerable amount of variability in annual density estimates, the past decade is characterized by some of the highest recorded densities in both the North Fork Coeur d'Alene and St. Joe rivers. In particular, increased densities of Westslope Cutthroat Trout ≥300 mm reflect substantial improvements in size structure. We continue to see increases in Mountain Whitefish densities in the lower portions of the Coeur d'Alene and St. Joe rivers. Rainbow Trout densities remain at extremely low abundance throughout the St. Joe and North Fork Coeur d'Alene rivers. We continued to document relatively high densities of Rainbow Trout in the Little North Fork Coeur d'Alene River; notwithstanding, Westslope Cutthroat Trout densities also remain high in the Little North Fork Coeur d'Alene River. Rainbow Trout are known to compete and hybridize with Westslope Cutthroat Trout and the IDFG manages for low abundance of Rainbow Trout in the Spokane River Basin to reduce the potential for such interactions. The recent increase in density of Rainbow Trout in the Little North Fork Coeur d'Alene does not correspond to an increase in other portions of the basin, and is not currently a major management concern.

In recent history, a major concern among the angling public has been about the effect of summer conditions and its interaction with angling-induced fish mortality. We have continued to document that severe drought conditions during 2015 did not cause substantial direct mortality of Westslope Cutthroat Trout. Population density did decline in 2016, suggesting that some drought-induced mortality may have occurred. However, density has subsequently increased, suggesting that any mortality that may have occurred resulted in relatively minor and short-lived population impacts. Any mortality that may have resulted from drought conditions was likely buffered by a compensatory response. Flow conditions were closer to mean base flows in 2018, and again we did not observe any dead Westslope Cutthroat Trout at any of our snorkel sites nor did we receive public comments about dead fish being observed during the summer and early-fall months.

Although anecdotal, such observations might indicate a positive relationship between extreme summer conditions and Westslope Cutthroat Trout mortality. Both river systems displayed similar total Westslope Cutthroat Trout density to 2017; current densities were near the 10-year mean, and above the historical mean in both river systems. The long-term effects of severe summer drought conditions on recruitment dynamics and somatic growth are not yet understood, but will probably be revealed through continued annual monitoring.

RECOMMENDATIONS

1. Continue to monitor wild trout abundance and population characteristics in the upper Spokane River Basin.

2. Continue to monitor trends in fish assemblage characteristics.

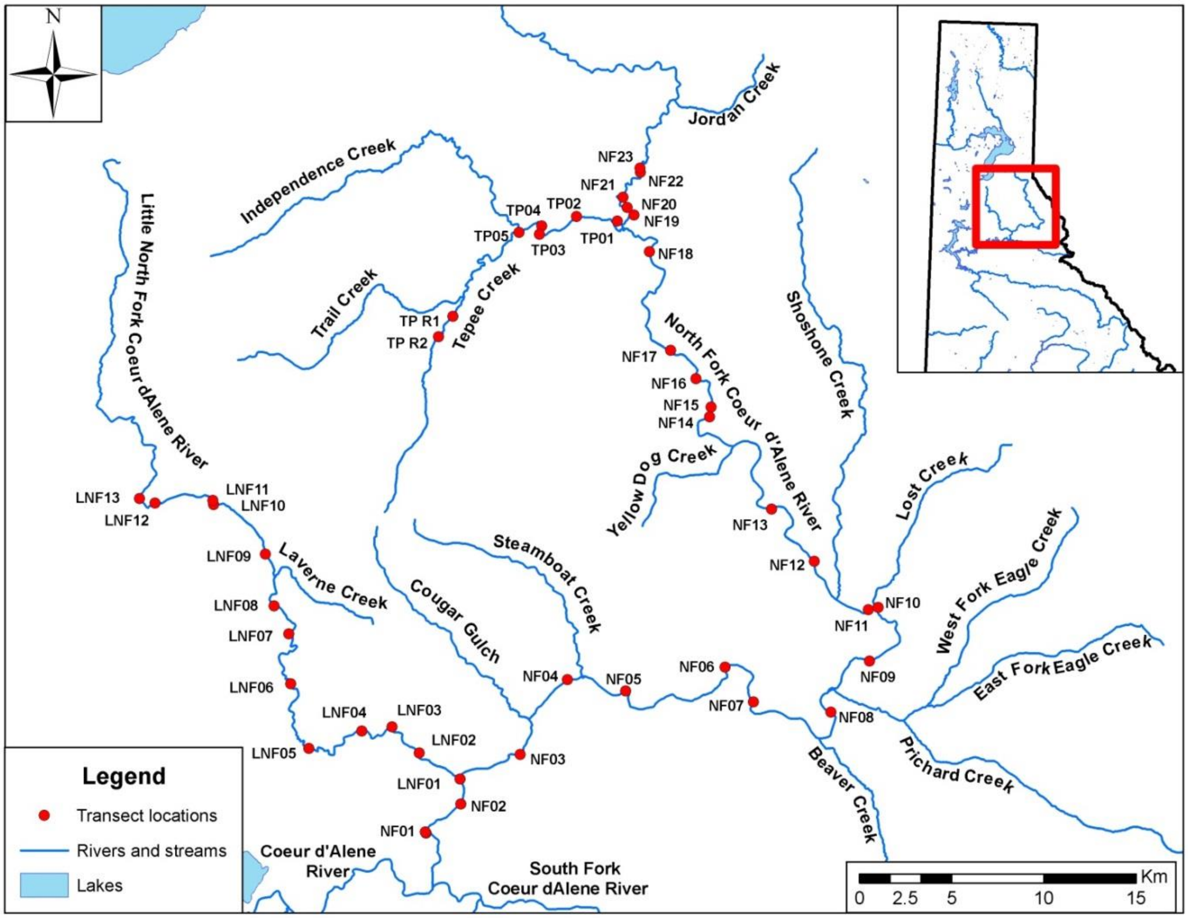

Figure 46. Location of 44 index reaches sampled using snorkeling in the Coeur d'Alene River, Idaho during July 30–31, 2018.

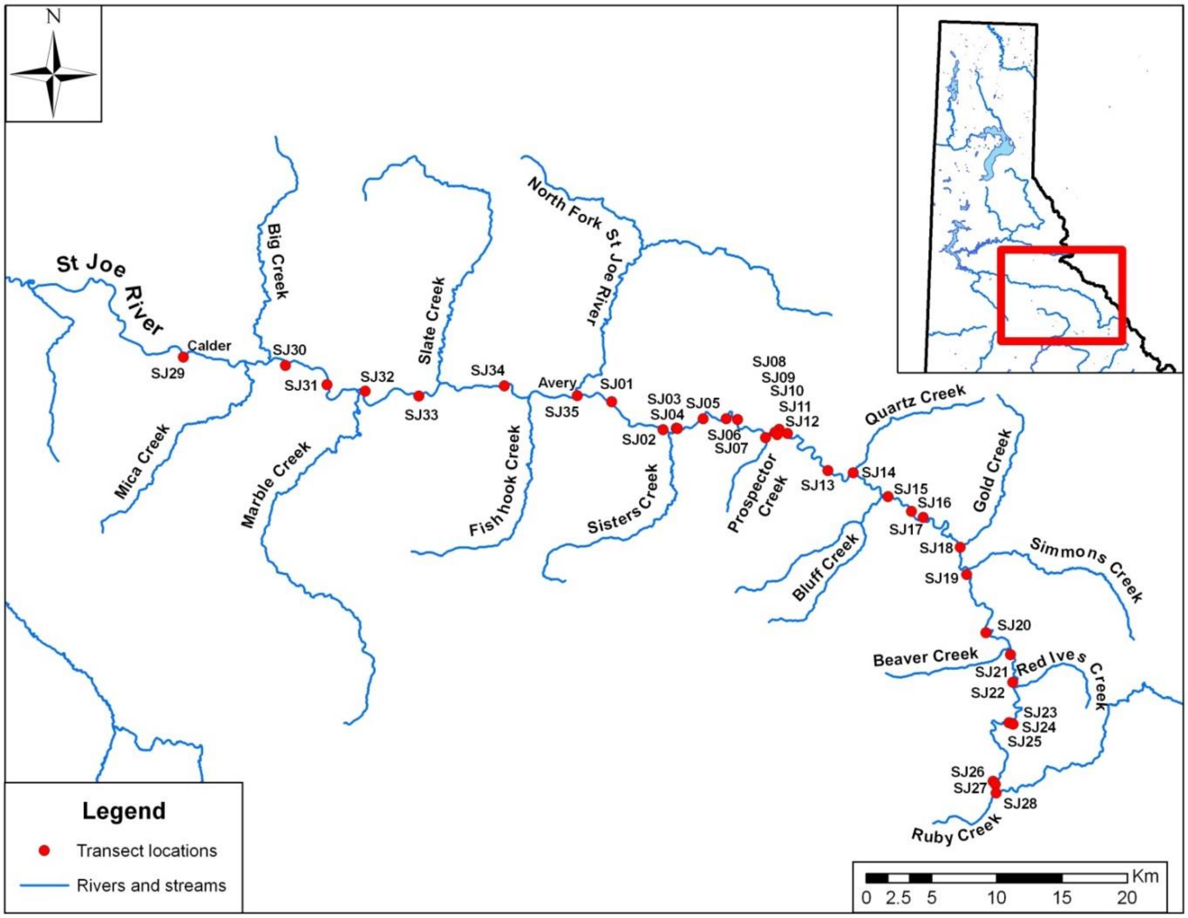

Figure 47. Location of 35 index reaches sampled using snorkeling in the St. Joe River, Idaho during August 7–8, 2018.

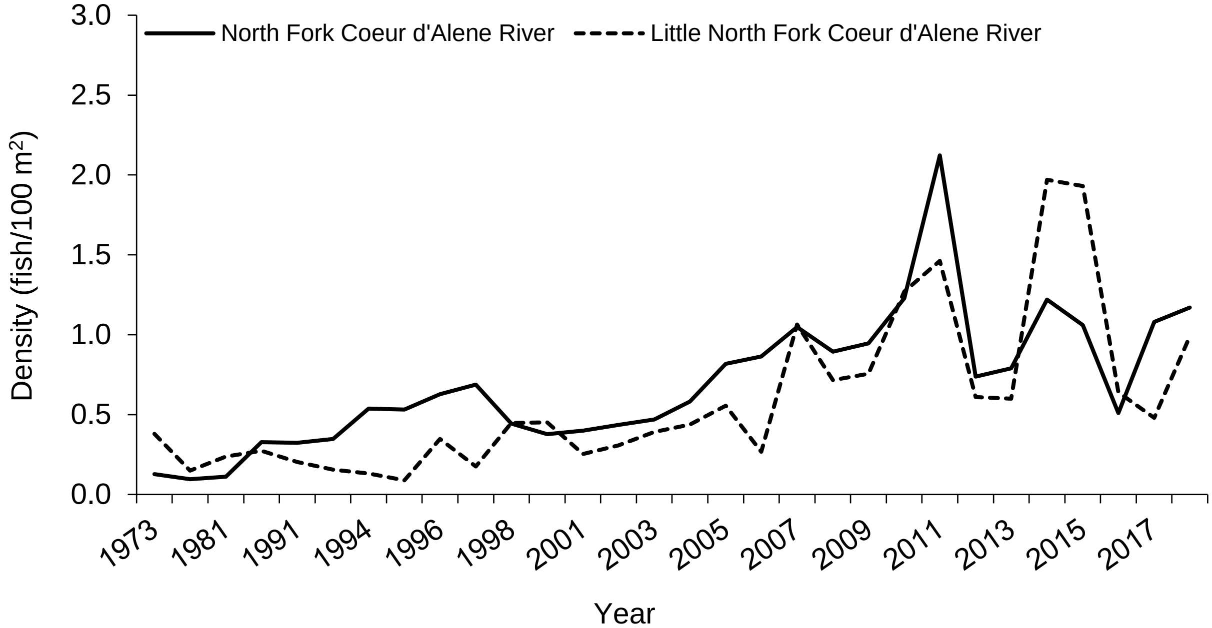

Figure 48. Mean density of Westslope Cutthroat Trout observed during snorkeling in the North Fork of the Coeur d'Alene River and Little North Fork of the Coeur d'Alene River (1973–2018).

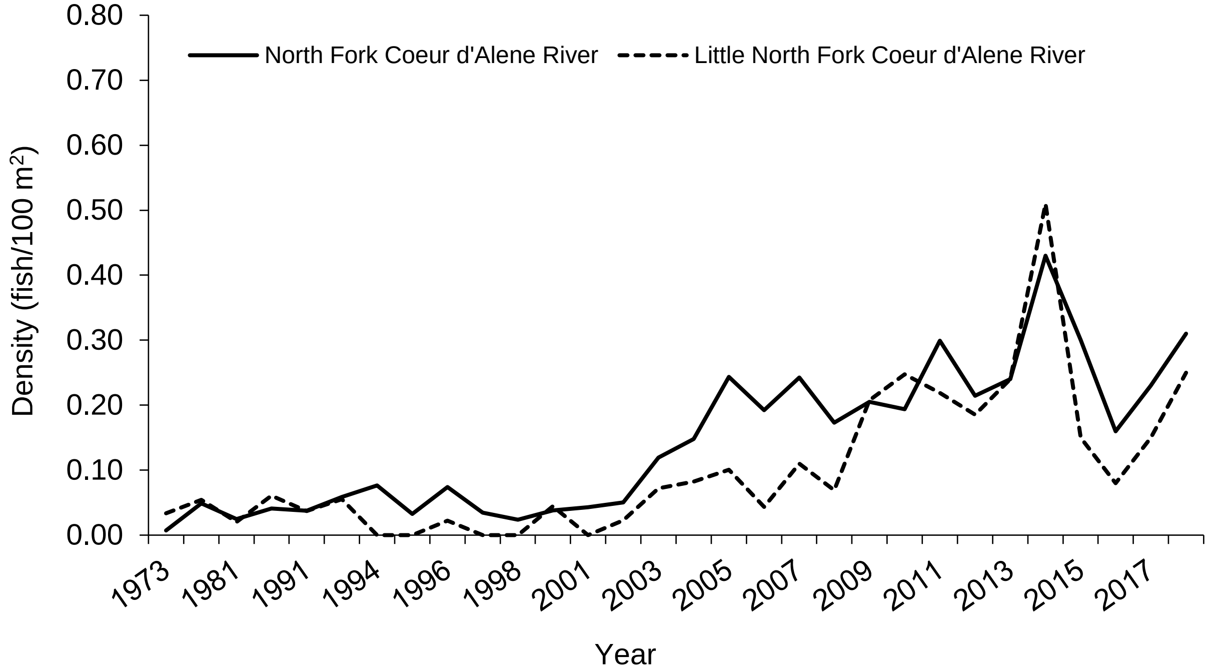

Figure 49. Mean density of Westslope Cutthroat Trout larger than 300 mm TL observed during snorkeling in the North Fork of the Coeur d'Alene River and Little North Fork of the Coeur d'Alene River (1973–2018).

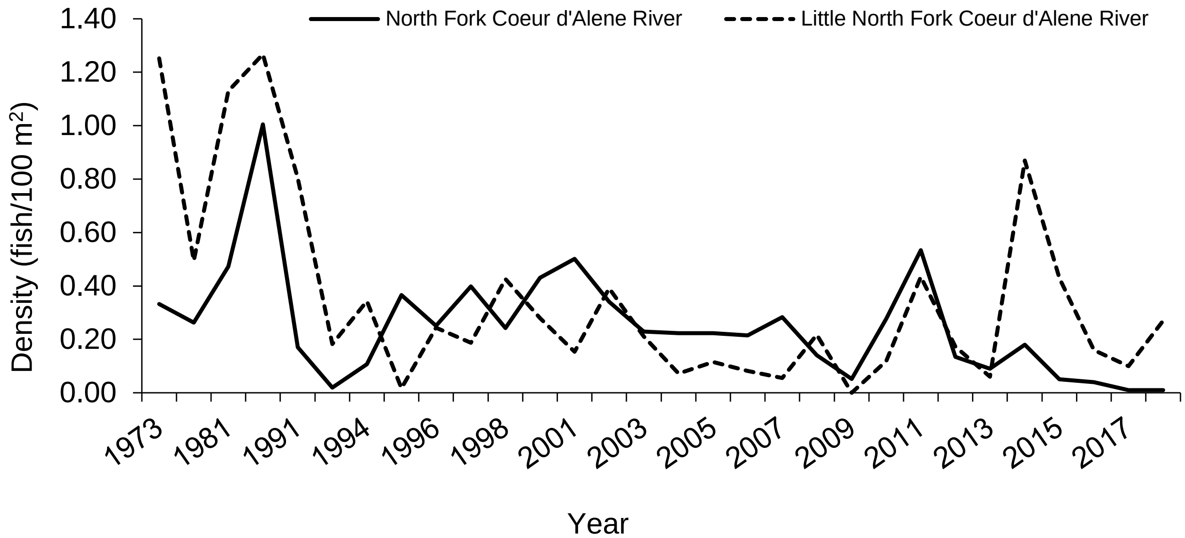

Figure 50. Mean density of Rainbow Trout observed during snorkeling in the North Fork of the Coeur d'Alene River and Little North Fork of the Coeur d'Alene River (1973–2018).

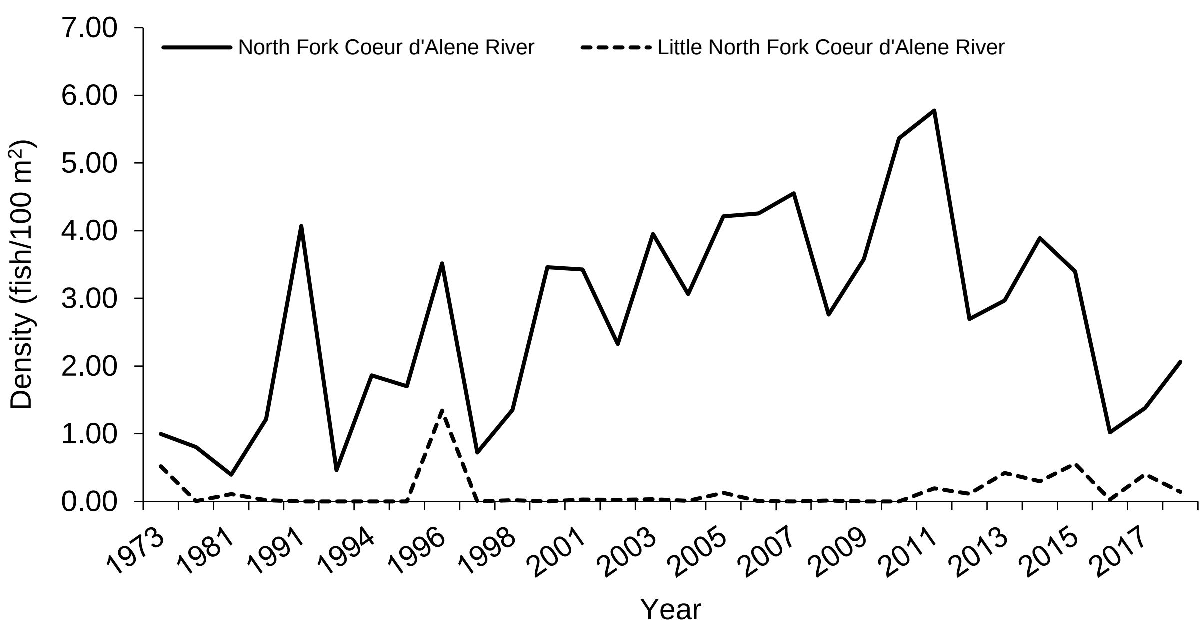

Figure 51. Mean density of Mountain Whitefish observed during snorkeling in the North Fork of the Coeur d'Alene River and Little North Fork of the Coeur d'Alene River (1973–2018).

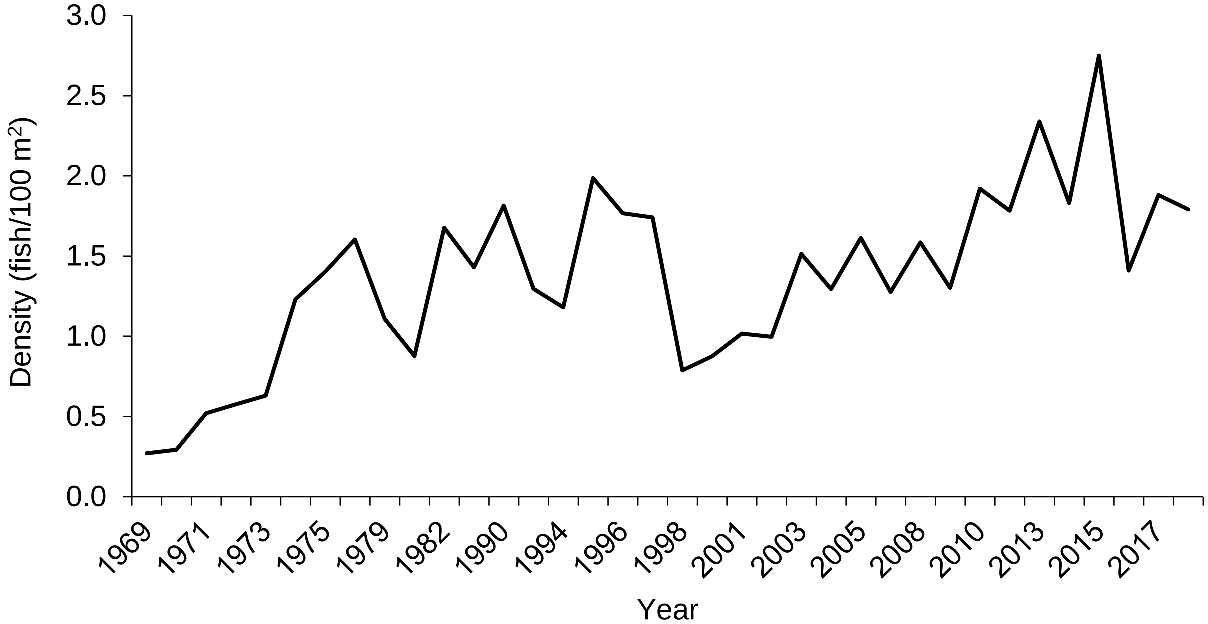

Figure 52. Mean density of Westslope Cutthroat Trout observed during snorkeling in the St. Joe River (1969–2018).

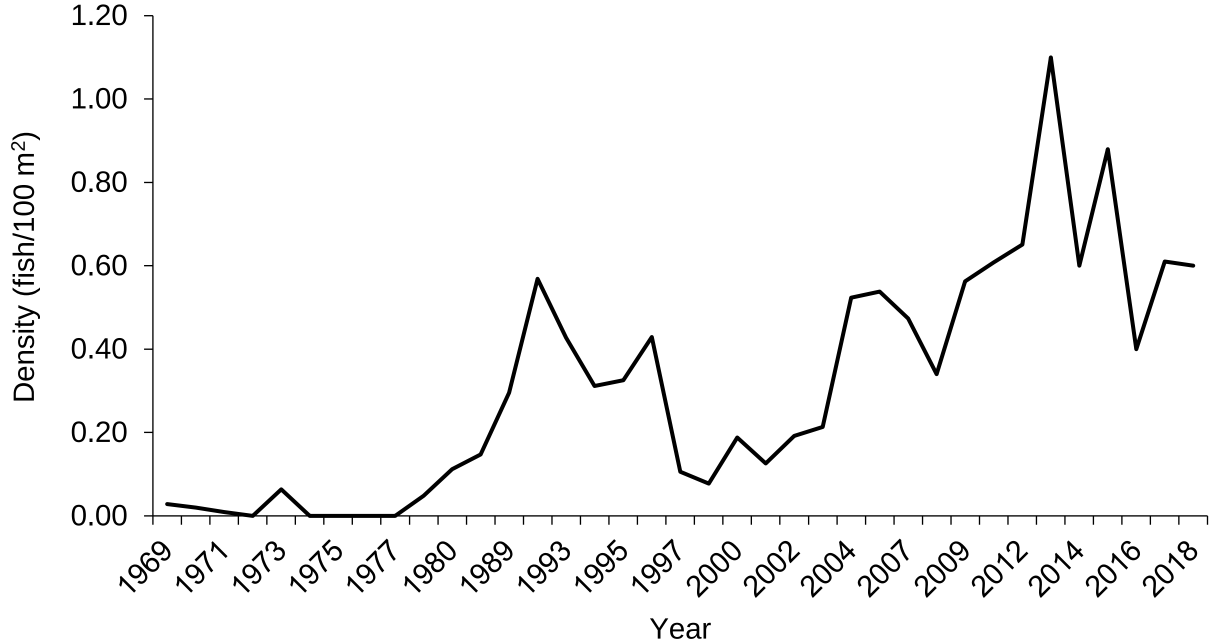

Figure 53. Mean density of Westslope Cutthroat Trout larger than 300 mm TL observed during snorkeling in the St. Joe River (1969–2018).

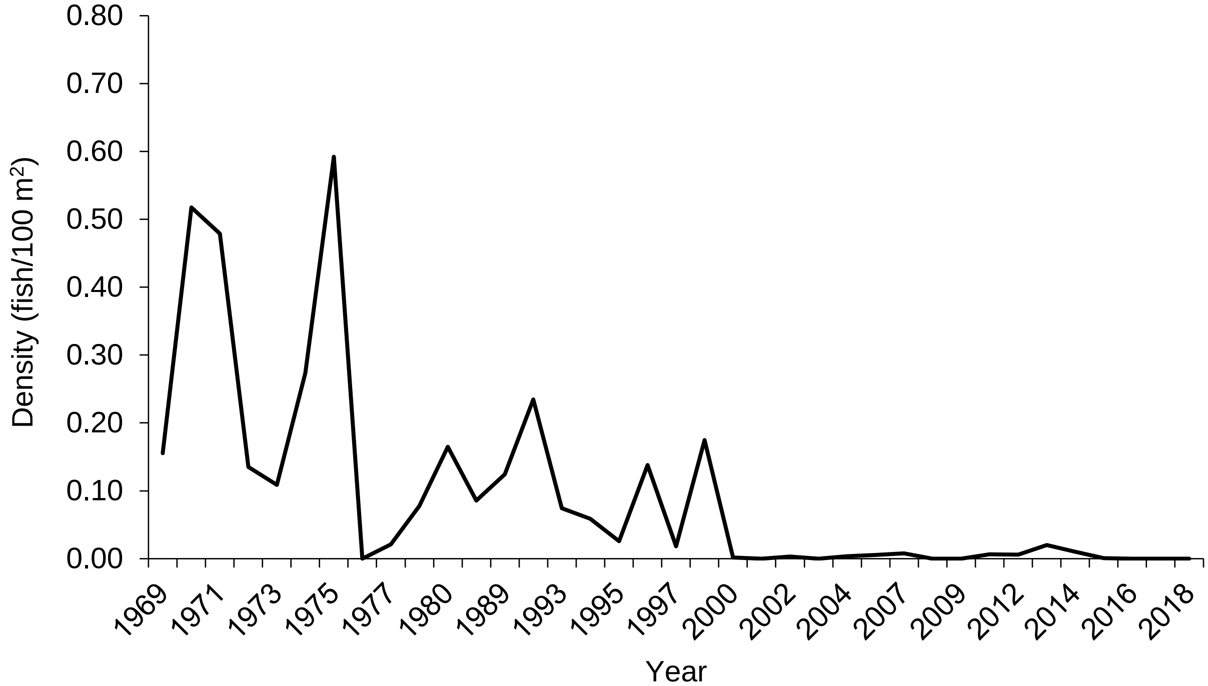

Figure 54. Mean density of Rainbow Trout observed during snorkeling in the St. Joe River (1969–2018).

Figure 55. Mean density of Mountain Whitefish observed during snorkeling in the St. Joe River (1969–2018).

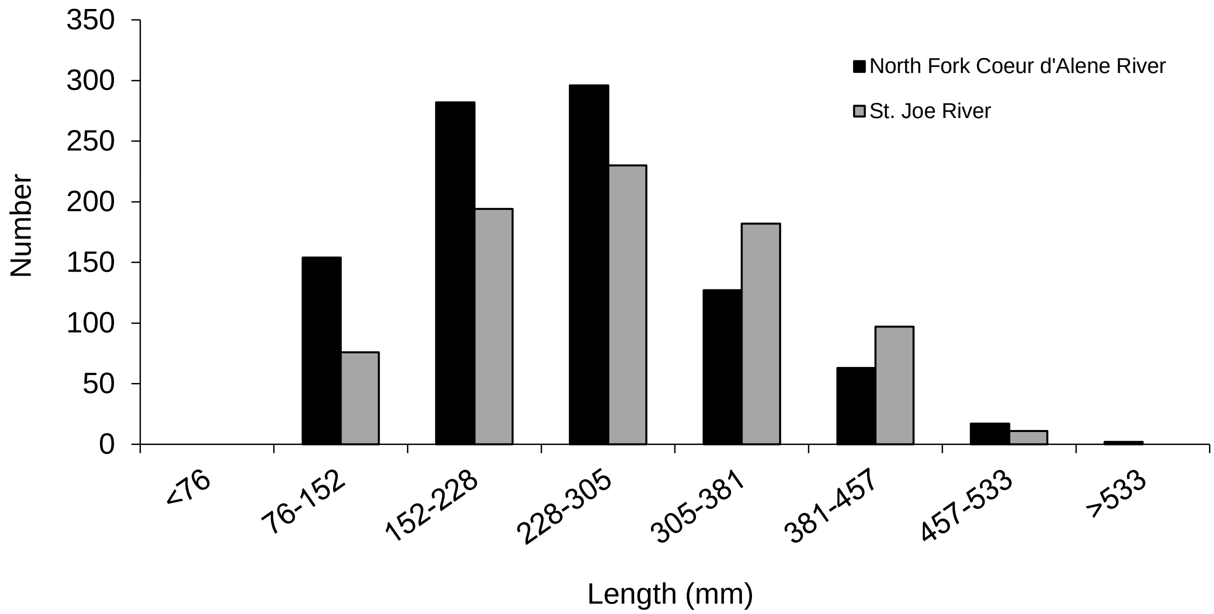

Figure 56. Length-frequency distributions of Westslope Cutthroat Trout observed during snorkeling in the North Fork Coeur d'Alene River (includes Little North Fork Coeur d'Alene River and Teepee Creek) and St. Joe River (2018).

BRICKEL CREEK AND FISH CREEK FISH ASSEMBLAGE STRUCTURE

ABSTRACT

We conducted fish assemblage and habitat assessments in Brickel and Fish creeks to complement previous work focused on understanding fish community and population structure in the "sink" drainages surrounding the Rathdrum Prairie. In particular, we were interested in evaluating how these streams interact with the lentic environments (i.e., Spirit and Twin lakes) at their outlet. We surveyed nine total sites on Brickel (n = 5) and Fish (n = 4) creeks using backpack electrofishing and common stream habitat assessment techniques. We documented the presence of Brook Trout *Salvelinus fontinalis*, Cedar Sculpin *Cottus schitsuumsh*, Rainbow Trout *Oncorhynchus mykiss*, and Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi*. All fishes appeared to be of wild origin. Both streams are likely important for supporting some level of wild adfluvial salmonid production and contribute to fishery diversity in their parent lake. We observed little evidence of angling activity and surmise that no significant local fisheries are supported in either stream, although limited Brook Trout angling opportunity exists. Physical channel characteristics were similar between streams, but we observed differences in instream and terrestrial cover and substrate composition. Habitat information suggests that land use activities probably influence stream habitat characteristics, but in different ways between the two drainages. We found that sedimentation is more prevalent in the Fish Creek drainage and that early successional riparian vegetation was more common in Brickel Creek and currently limits stream shading and fish cover. Fishery managers should be aware that unique opportunities may arise in the future to partner with private land managers in the Brickel Creek drainage to improve or maintain important habitat for adfluvial Rainbow Trout.

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INTRODUCTION

The area surrounding the Rathdrum Prairie in northern Idaho contains several isolated "sink" tributaries to the Spokane and Pend Oreille rivers. Those tributaries originate in the Coeur d'Alene and Selkirk mountains along the east and west extents of the Rathdrum Prairie, and discharge from those systems is subterraneous at the valley floor. Although isolated through geologic processes, these streams have been historically occupied by native fluvial species including Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* (IDFG 2013). Some anecdotal information has suggested that Brook Trout *Salvelinus fontinalis* are currently established in many of these tributaries (IDFG 2013). In 2013, Ryan et al. (2014) surveyed three sink drainages within the Spokane River drainage (i.e., Lewellen, Sage, and Lost creeks) to evaluate fish assemblage characteristics. The authors reported presence of Westslope Cutthroat Trout and establishment of naturalized populations of Brook Trout. The information resulting from this assessment has furthered our understanding of fisheries in the Panhandle Region, but also provided specific information on the role these streams play in Westslope Cutthroat Trout conservation.

For this assessment, we sought to complement the survey from Ryan et al. (2014) by providing additional empirical information on fish assemblages in sink tributaries around the Rathdrum Prairie. Our inventory focused on Brickel and Fish creeks, which represent the primary tributaries to Spirit and Twin lakes, respectively. Brickel and Fish creeks are unique among the sink drainages because these tributaries interact with a lentic system. Lowland lake surveys conducted on Spirit Lake during 2016 showed evidence that adfluvial Rainbow Trout and Brook Trout likely occur in that system (Ryan et al. 2020b). As such, a major focus of this work was to understand the importance of each tributary for adfluvial trout and, more specifically, to answer questions about how those tributaries function to support the fishery in Twin and Spirit lakes.

OBJECTIVES

1. Characterize fish assemblage structure in Brickel and Fish creeks.

2. Estimate stock structure of sport fish species in Brickel and Fish creeks.

3. Describe habitat in Brickel and Fish creeks.

4. Evaluate fish assemblage- and species-level relationships with habitat.

METHODS

Fishes were sampled from Brickel and Fish creeks during July – August of 2017 when stream discharge permitted safe wading. The mainstem length of each stream existing within the state of Idaho (both drainages are interstate) was measured and sample locations were then selected systematically at 3-km intervals between the inlet and ID-WA boundary. Near the location of each site, reaches were identified based on major macrohabitat transitions between riffles extending between 100–200 m in length. Backpack electrofishing was used to capture fishes. Electrofishing equipment consisted of a Smith-Root model LR-24 electrofisher (Smith-Root, Inc., Vancouver, Washington, USA) using pulsed-DC current set to 600–800 v and 40–50 hz depending on water conductivity and temperature. During sampling, one person operated the electrofishing equipment and two netters collected immobilized fish adjacent to the operator. Sampling consisted of a single upstream electrofishing pass, beginning and ending at transitions to riffle macrohabitats. Upon completion of each reach, all fishes were identified to species and measured for total length (TL). Surface area (m^2) was estimated at each site to provide a measure of sampling effort. The number of each species observed was divided by the surface area sampled to provide a standardized relative abundance measure (DuPont et al. 2009).

Habitat information was collected to understand the influence of abiotic factors on fish assemblage structure and species relative abundance. Depth, wetted width, substrate composition, bank type, and woody debris were measured within each reach following fish sampling. Habitat sampling transects were established at 10-m intervals along each reach and instream variables measured at 1-m^2 areas around five equidistant points along each transect. Depth was measured to the nearest 0.1 m, and substrate was visually estimated as the proportion belonging to one of five categories: silt–sand (< 0.0004–0.2 mm), gravel (0.2–64.0 mm), cobble (64.0–256.0 mm), boulder (> 256.0 mm), and bedrock (modified from Orth and Maughan 1982). The proportion of both banks belonging to the following four categories was also visually estimated: eroding, vegetated, silt–sand (0.2 mm), and cobble–boulder (i.e., riprap structure; 64.0 mm). The amount of woody debris was calculated as the total surface area of woody instream cover that was greater than 0.2 m in diameter and greater than 0.5 m in length (Watkins et al. 2015).

RESULTS

We sampled a total of 167 fish at five sites in Brickel Creek and 55 fish at the four sites in Fish Creek during July 5–7, 2017. Fish were detected at all reaches in both streams and the most abundant species was Cedar Sculpin. Brook Trout and Cedar Sculpin were detected at all sites in both streams and neither species appeared to show patterns in relative abundance as a function of lake proximity. Westslope Cutthroat Trout were the least abundant species detected in Brickel and Fish creeks. Relative abundance of Westslope Cutthroat Trout was slightly higher in Brickel Creek and the species was detected at three sites in Brickel Creek and one site in Fish Creek. Rainbow Trout were only present in Brickel Creek and were well-distributed throughout the system. Size structure of salmonids in both streams was poor and few stock length and larger individuals were sampled. Estimates of size structure characteristics and population densities can be found in Table 39 and Figures 57–60.

The study streams showed variable patterns in terms of habitat structure and longitudinal variability. Brickel Creek has a slightly larger watershed which contributes greater annual discharge, but the physical stream channel characteristics of the two watersheds are otherwise comparable. Mean wetted stream width was slightly higher across sites in Brickel Creek, but mean water depth and its associated variance was similar between streams (Table 40). The greatest differences we observed were for substrate composition and for instream and terrestrial cover. Individual estimates of substrate composition were pooled (see Table 41) to provide a general understanding of the relative proportion of fine and coarse substrate within and between streams. In general, Brickel Creek had a higher mean proportion of coarse substrates with more fine substrates observed in the farthest downstream site. Fish Creek also showed a pattern of increasing substrate size as a function of outlet proximity, but exhibited higher proportions of fine substrates across all sites. Estimates of instream and terrestrial cover were higher in Fish Creek across all sites. Within sites, instream and canopy cover estimates tended to show some correlation whereby sites with relatively low instream cover were often associated with low canopy cover. Longitudinal patterns in cover were not evident in study streams and cover tended to exhibit high variability among and within sties.

DISCUSSION

This study was largely motivated by a desire to understand how sink tributaries on the western Rathdrum Prairie interact with their parent lake environments. In particular, Spirit Lake supports a wild-origin Rainbow Trout fishery; however, an understanding of recruitment sources and the interaction between the lake and its tributaries was formerly not understood. Anecdotal information from the public suggests that neither Brickel nor Fish creeks support significant fisheries in and of themselves. However, some local anglers have noted that much of the angler effort is likely focused on Brook Trout. As such, fishery management activities in Brickel and Fish creeks that improve recruitment for adlfuvial salmonid populations could be beneficial for enhancing fisheries in Spirit and Twin lakes.

The stock structure of salmonids we observed in both streams likely reflects that of populations having poor growth of fluvial residents, or more likely, is comprised almost entirely of juvenile adfluvial migrants. We documented wild Westslope Cutthroat Trout in both streams which are likely relicts of native populations and naturalized hatchery fish. Westslope Cutthroat Trout tended to be in very low abundance throughout both streams, and the associated lake fisheries for Westslope Cutthroat Trout are probably most strongly supported by current stocking activities. This is especially true in Spirit Lake where a troll fishery is partially supported by hatchery Westslope Cutthroat Trout stockings (fingerlings) on an annual basis. Westslope Cutthroat Trout have not been stocked in Lower Twin Lake since 2014, yet a recent lowland lake survey documented what were presumably wild juveniles. Relative abundance of Westslope Cutthroat Trout in Lower Twin Lake was below what we think could reasonably support a fishery; however, we think that wild production from Fish Creek may be important for diversifying the troll fishery. Similarly, Brook Trout populations in both streams appear to be the derivative of an adfluvial life history based on recent lowland lake survey information (Ryan et al. 2020b). This has not been verified and we acknowledge that Brook Trout often spawn successfully in lake margins, but this tends to only occur when suitable fluvial habitat is nonexistent (Koenig 2012).

The most significant risk to adfluvial salmonids in Brickel and Fish creeks is habitat degradation from land use and recreation (Rankel 1971; Shepard et al. 2005). Both drainages are almost entirely owned by timber management corporations and actively managed for timber harvest. As such, sedimentation associated with logging activities, road building, and vehicle traffic is likely to impair instream habitat for salmonids. No historical fish survey data are available for either stream, but we assume that stream habitat improvement measures (i.e., Forest Practices Act regulations and best timber management strategies) have improved or maintained the terrestrial component of habitat. In fact, the benefit of intact riparian buffers for maintaining stream canopy cover was visually evident in our survey sites where clear-cutting had occurred nearby—nearly all sites with clear-cuts in close proximity had higher instream and canopy cover estimates and coarse substrate composition than those without recent adjacent clear-cuts. With overall respect to the influence of land use in the two drainages, the amount of active timber harvest and distribution of roads appears to be similar, and the existing habitat is suitable for supporting salmonid populations to some degree. Our observations show that sedimentation may be a more imminent concern in the Fish Creek drainage with regard to maintaining substrate composition that is favorable for salmonids. In Brickel Creek, stream shading from riparian vegetation and instream cover provided by terrestrial inputs are limited due to the immaturity of riparian communities. As such, succession of riparian communities in the Brickel Creek drainage has the potential to improve recruitment potential for adfluvial salmonids and subsequently benefit the Spirit Lake trout fishery.

RECOMMENDATIONS

1. Where appropriate, collaborate with private timber management organizations to identify opportunities to improve salmonid habitat in Brickel Creek.

Table 39. Sample size (*n*), total length (mm; Minimum–Maximum [Min–Max]) statistics, and density (fish/100 m^2) for fish populations sampled from Brickel and Fish creeks (2017). Numbers in parentheses represent one standard error about the mean.

		Total length		
Species	n	Mean	Min–Max	Density
		Brickel Creek		
Brook Trout	25	100.8 (10.8)	46–225	1.0 (0.2)
Cedar Sculpin	81	62.1 (2.0)	33–113	3.3 (1.1)
Rainbow Trout	52	78.3 (4.5)	27–135	2.1 (0.6)
Westslope Cutthroat Trout	9	124.7 (22.5)	80–298	0.4 (0.3)
		Fish Creek		
Brook Trout	26	122.2 (12.8)	43–311	1.4 (0.5)
Cedar Sculpin	26	68.0 (3.3)	36–92	1.5 (0.2)
Westslope Cutthroat Trout	3	113.7 (17.8)	92–142	0.2 (0.2)

Table 40. Mean estimates (SEs in parentheses) of the habitat variables measured at sampling sites in Brickel and Fish creeks (2017). Sites are organized numerically in upstream ascending order.

	Site				
Variable	1	2	3	4	5
			Brickel Creek		
Width	5.7 (0.5)	6.5 (0.4)	5.7 (0.4)	4.2 (0.3)	4.2 (0.5)
Depth	0.4 (0.05)	0.2 (0.02)	0.3 (0.02)	0.3 (0.04)	0.2 (0.02)
$\text{Substrate}_{\text{Fine}}$	33.6 (5.2)	20.5 (5.4)	7.8 (2.5)	18.6 (3.4)	17.0 (3.5)
$\text{Substrate}_{\text{Coarse}}$	66.4 (5.2)	79.5 (5.4)	92.2 (2.5)	81.4 (3.4)	83.0 (3.5)
$\text{Cover}_{\text{Instream}}$	16.0 (5.4)	2.0 (1.1)	1.4 (0.5)	6.9 (1.9)	10.1 (2.6)
$\text{Cover}_{\text{Canopy}}$	13.8 (3.1)	9.0 (1.2)	8.0 (1.6)	22.1 (5.5)	15.0 (3.0)
			Fish Creek		
Width	4.3 (0.5)	5.1 (0.7)	4.9 (0.4)	3.6 (0.3)	--
Depth	0.4 (0.02)	0.2 (0.02)	0.2 (0.02)	0.2 (0.01)	--
$\text{Substrate}_{\text{Fine}}$	56.0 (10.2)	41.1 (6.0)	32.2 (4.7)	40.0 (4.7)	--
$\text{Substrate}_{\text{Coarse}}$	44.0 (10.2)	58.9 (6.0)	67.8 (4.7)	60.0 (4.7)	--
$\text{Cover}_{\text{Instream}}$	15.3 (3.3)	18.3 (6.0)	26.7 (7.0)	21.5 (2.6)	--
$\text{Cover}_{\text{Canopy}}$	28.5 (5.0)	16.4 (2.9)	46.7 (12.1)	21.0 (4.3)	--

Table 41. Descriptions of habitat variables summarized to assess abiotic conditions in Brickel and Fish creeks during the summer of 2017.

Variable	Description
Width	Mean wetted width (m)
Depth	Mean water column depth (m)
$Substrate_{Fine}$	Proportion of substrate (%) consisting of fine particles (≤2 mm diameter)
$Substrate_{Coarse}$	Proportion of substrate (%) consisting of coarse particles (≥64 mm diameter)
$Cover_{Instream}$	Proportion of overhead submerged cover (%) provided by large substrate or woody debris
$Cover_{Canopy}$	Proportion of wetted transect width with overhanging vegetation

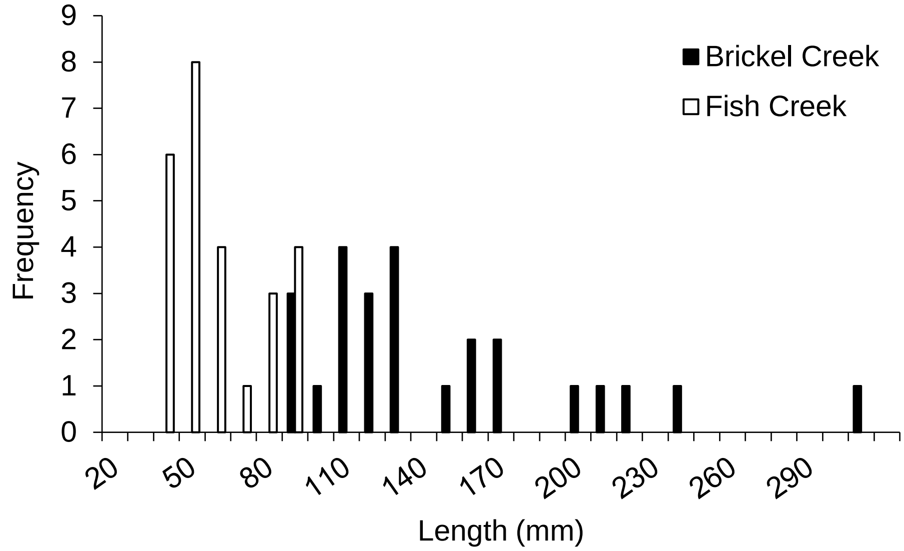

Figure 57. Length-frequency distributions of Brook Trout populations sampled from Brickel and Fish creeks (2017).

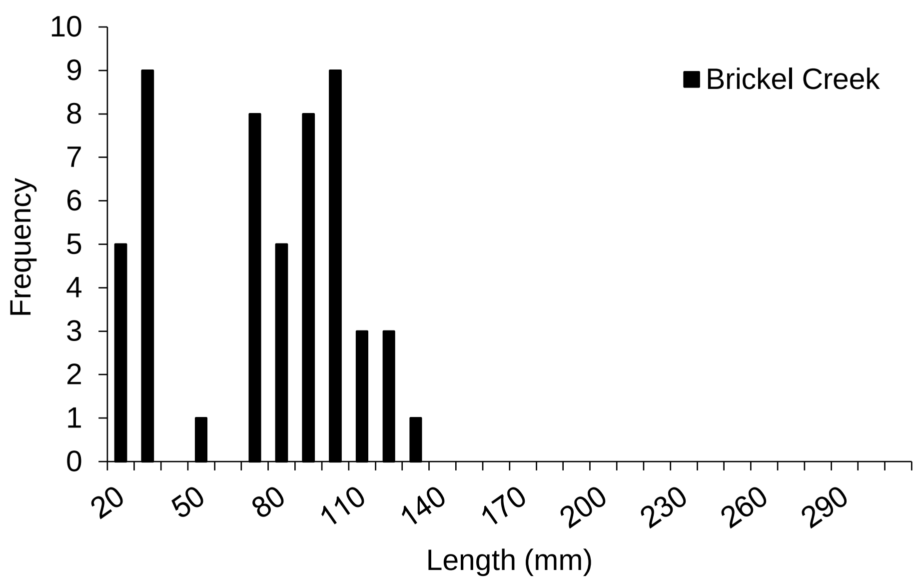

Figure 58. Length-frequency distribution of the Rainbow Trout population sampled from Brickel Creek (2017).

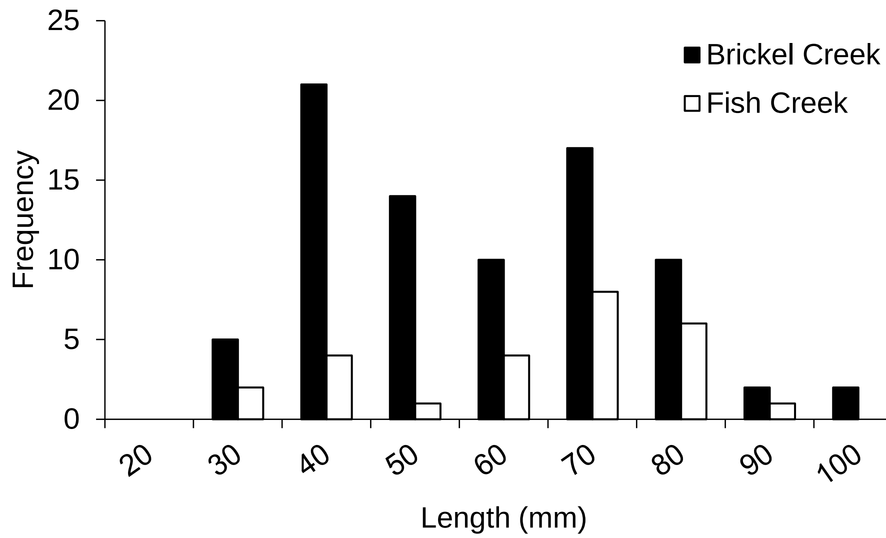

Figure 59. Length-frequency distributions of Cedar Sculpin populations sampled from Brickel and Fish creeks (2017).

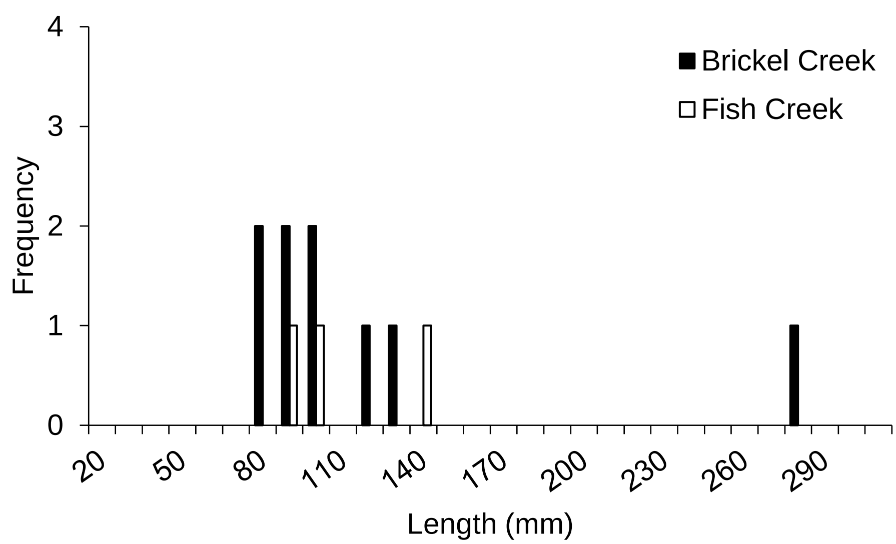

Figure 60. Length-frequency distributions of Westslope Cutthroat Trout populations sampled from Brickel and Fish creeks (2017).

SPIRIT LAKE CREEL SURVEY

ABSTRACT

We conducted a roving-access creel survey during April 2018 through March 2019 to understand angler dynamics on Spirit Lake in northern Idaho. Data were collected during April 2018–March 2019. Anglers fished an estimated 45,235 hours and reported catching 10 species during the surveyed period. Kokanee and Largemouth Bass were the most targeted species of fish. However, a majority of anglers fishing Spirit Lake were generalists and did not specifically target any one species contrary to the kokanee centric fishery from previous creel surveys. Catch rates varied widely by species and season.

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INTRODUCTION

Spirit Lake is located in Kootenai County near the town of Spirit Lake, Idaho. The lake has a surface area of approximately 596 hectares, a mean depth of 11.4 m, and a maximum depth of approximately 30 m. The lands surrounding the lake are primarily private ownership and many residences are located near the lakeshore. Public boating access to the lake can be found at an Idaho Department of Fish and Game (IDFG) access site on the northeast side of the lake, a Kootenai County access site on the east side of the lake, and second Kootenai County access site on the northwest side of the lake (no parking available).

Spirit Lake is managed under general regional fishing regulations, including daily bag limits of six trout (all species combined) and six bass (both species combined). The only special regulation is for kokanee, which allows a daily bag of 25 fish instead of the general limit of 15 fish. Westslope Cutthroat Trout *Oncorhynchus clarkii* have been stocked annually since 1994 and Fall Chinook have been stocked annually since 2016. Rainbow Trout *Oncorhynchus mykiss* were stocked historically until 1994 at both catchable length (i.e., 152 - 305 mm) and fingerling length (i.e., 76 – 152 mm). Brook Trout *Salvelinus fontinalis* were also stocked in the drainage twice in the early-1980s. Both Rainbow Trout and Brook Trout now persist through natural recruitment (Ryan et al. 2020b). Other species previously identified in Spirit Lake include Black Crappie *Pomoxis nigromaculatus,* Bluegill *Lepomis macrochirus,* Brown Bullhead *Ameiurus nebulosus,* Largemouth Bass *Micropterus salmoides*, Mountain Whitefish *Prosopium williamsoni*, Pumpkinseed *Lepomis gibbosus,* Smallmouth Bass *Micropterus dolomieu,* and Yellow Perch *Perca flavescens* (Ryan et al. 2020b).

Understanding the way that anglers interact with fish populations is central to fisheries management for a variety of reasons. Information about how anglers behave and characterizing their experiences provides managers with a basis for guiding the management of a fishery. Thus, our objective was to describe the fishery use, quality and angler preferences using a year-long creel survey.

METHODS

We conducted a year-long creel survey on Cocolalla Lake from April 2018 through March 2019 using a roving-access design (Pollock et al. 1994). Survey design and analysis was completed using a customized creel survey database (Josh McCormick, IDFG, personal communication). The survey period was divided into two-week intervals during April through September. One-month intervals were used during October through March. Intervals were stratified by day type, including weekdays and weekend/holidays. We scheduled four survey days per interval, including two weekdays and two weekend/holidays. Survey dates were randomly chosen. Daily start times for an eight hour survey shift were also randomly chosen. We coordinated the Cocolalla Lake survey with a concurrent survey of Spirit Lake. Eight hour shifts were divided into two four-hour periods, one period per fishery. The first period was alternated between fisheries.

Roving counts of boats and shore anglers were conducted twice per shift at randomly scheduled times to estimate angler effort. Creel clerks made a single loop around the lake by boat for each scheduled count. During periods when the lake was iced covered, ice anglers were counted by walking out on the ice at the three access sites. Ice anglers were counted as shore anglers for the purpose of survey analysis.

Angler interviews were conducted to obtain catch rate data and describe angler type. Interviews were completed at the IDFG access site along Maine Street on the northeast end of the lake . Creel clerks waited at the access site to intercept anglers leaving the lake upon completion of their angling effort. We attempted to interview all angling parties leaving through the access site during the survey period. Interview questions included number of anglers, angler type (boat or shore), number of rods fished, time spent fishing, targeted species, number of fish kept per species, number of fish released per species, and whether a daily trip was completed.

Daily fishing effort was first estimated for each day within a sampling interval for which a survey was completed. Daily fishing effort was estimated as average angler count multiplied by the number of possible fishing hours in the sampled day. Fishing hours were described as the period between sunset and sunrise. Daily fishing effort was expanded to the temporal strata (i.e., two-week period or month and day type) by dividing by the sampling probability:

$$E = e/pt,$$

where E = total effort, e = sampling period effort (daily fishing effort), and pt = temporal sampling probability. Sampling probabilities were estimated by day type as the number days sampled within a strata divided by the number of days within the strata. Fishing effort estimates by day type were then summed across strata for an estimate of total fishing effort by month.

Catch rate was reported as the number of fish caught, harvested, or released per angler hour. Catch rate was estimated from completed trip interviews and was calculated by the ratio of means estimator (Pollock et al. 1994). Total catch was divided by total angler effort for various hierarchies of the survey design (i.e., monthly or total). The total number of fish released, harvested, and caught (harvest + release) were estimated by multiplying total fishing effort by the appropriate total rate estimator (harvest, release, or catch) for the various hierarchies of the design. Total catch was estimated as the product of catch rate and effort for each strata.

RESULTS

Anglers fished an estimated 45,235 hours on Spirit Lake from April 1, 2018 through March 31, 2019. Fishing occurred from boats and the shore. However, a majority (67%) of fishing effort was attributed to boat anglers. Angler effort was bimodal with a peak in June (9,612 h) followed a second smaller peak in February when sufficient ice provided safe ice fishing opportunities (5,999 h; Figure 61). Ice cover in December and January limited boat access and shore angling, but did not provide safe access for ice anglers. As a result, angler effort was zero in December and January and overall effort during the winter was much lower than effort observed in the remainder of survey (Figure 62).

A majority of anglers fishing Spirit Lake were generalists and did not indicate they were specifically targeting any one species (Table 42). Of those anglers who did specify a targeted species, Largemouth Bass was their most sought after species. Kokanee was also targeted by a large proportion of anglers.

Catch rates varied widely by species and season (Tables 43 and 44). Anglers reported catching 13 species throughout the survey period. Angler catch rates for the entire study period were highest for Largemouth Bass (0.47 fish/h), and peak Largemouth Bass catch rates occurred during August (0.89 fish/h). Kokanee catch rates for the entire study period were nearly as high as Largemouth Bass. However, Kokanee catch rates were highest in February (2.28 fish/h) and were the highest for any species in any month. Total catch rates were similar between Rainbow Trout and Westslope Cutthroat Trout (0.06 and 0.04 fish/h, respectively). However, peak catch rates differed between the two species. Rainbow Trout catch rate were highest in July and peak catch rates for Westslope Cutthroat Trout occurred in February. Catch rates for panfish, such as Bluegill and Black Crappie, were highest in May and June.

Catch from the Spirit Lake fishery largely reflected targeted angler effort. Largemouth Bass and kokanee dominated the catch (Table 45). However, harvest rates varied by species. Nearly all Largemouth Bass were released (Table 45). Most kokanee were harvested and total catch during the short ice fishing season was similar to the remainder of the study period (Figure 63). Other species, such as Black Crappie, Rainbow Trout, Westslope Cutthroat Trout, and Yellow Perch did not substantially contribute to the overall catch and were not frequently targeted by anglers, but had moderate to high harvest rates when encountered. \

DISCUSSION

Spirit Lake has been and continues to be a popular fishery in the Panhandle region supporting both a summer and winter ice fishery. Angler effort is relatively high compared to other lakes in the region, such as Cocolalla Lake (19,733 angler hours during the same time period as our survey; see “Cocolalla Lake Investigations 2018” chapter of this report). Effort is typically bimodal with a peak in early summer and late winter. In early summer, water levels are up and provide the best boating access. The main access is the IDFG boat launch which is a shallower ramp that is difficult to launch boats in late summer and fall when water levels are lower. This likely reduces access to the fishery since the majority of anglers surveyed were boat anglers. In most years, sufficient ice formation allows for several weeks of ice fishing and expanded opportunities for shore anglers.

Historically, high densities of small kokanee have been immensely popular with ice anglers However, winter effort in our 2018 survey was substantially lower than the previous creel survey in 1999 (Fredericks et al. 2002). Mild winter weather conditions did not provide safe ice for fishing, except for a short time period in February. In addition, kokanee abundance was much lower than previous years (see “Lake Coeur d’Alene and Spirit Lake Kokanee Evaluations” chapter of this report) likely reducing harvest despite the increased daily bag limit from 15 to 25 kokanee adopted in 2016.

The fish community of Spirit Lake was described in a 2016 lowland lake survey as diverse with a robust warmwater component (Ryan et al. 2020b). Our 2018 creel survey reflected this diversity and deviated from previous creel surveys describing a kokanee-centric fishery (Davis et al. 1996; Fredericks et al. 2002). Generalist anglers targeting multiple species has doubled since 1992 likely taking advantage of the diverse fishing opportunities (Davis et al. 1996). Spirit Lake produces some of the largest Largemouth Bass in the Panhandle Region and anglers exclusively pursuing Largemouth Bass in Spirit Lake have quadrupled since 1992 (Davis et al. 1996; Maiolie et al. 2011; Ryan et al. 2018). Catch of naturalized Rainbow Trout and stocked Westslope Cutthroat Trout has also increased since previous creel surveys (Davis et al. 1996) and likely provide alternatives for summer trolling anglers when kokanee fishing is poor. Even though catch for Westslope Cutthroat Trout is lower than Rainbow Trout, stocking fingerling Westslope Cutthroat Trout into Spirit Lake is fairly inexpensive and it provides one of the few opportunities to harvest Westslope Cutthroat Trout in the region. Stocked Chinook Salmon were not encountered during the creel survey and were likely too small to recruit to the fishery, but should also provide another opportunity for trolling anglers in the future. We recommend management of the Spirit Lake fishery to focus on maintaining a diverse fish community to support current angler interests and alternatives when the kokanee fishery is poor.

Our angler survey design had limitations that may have influenced our results. For example, we only interviewed anglers at the IDFG access site. As such, anglers accessing the lake from alternative locations, such as private residences, were not included in our survey. While we made the assumption that angler catch rates did not vary by where anglers accessed the fishery, we did not test our assumption. We also experienced difficulty completing roving counts of anglers during the winter ice fishery and relied on available viewpoints to make counts. In addition, ice anglers commonly used shelters while fishing and made it difficult to count individual anglers. While we made an attempt to confirm angler counts during completed trip interviews, not all anglers completed their trip during a survey period. This limitation had the potential to underestimate angler effort during the winter ice fishery period.

RECOMMENDATIONS

1. Focus management of the Spirit Lake fishery on maintaining a diverse fish community to support the current diversity of angler interests.

2. Continue stocking Westslope Cutthroat Trout fingerlings.

3. Periodically conduct creel surveys to evaluate fishery performance.

Table 42. Proportion of angler reported targets by species or species group from anglers interviewed at Spirit Lake, Idaho from April 1, 2018 through March 31, 2019.

Species	% Anglers Targeting
Bass	32%
Black Crappie	1%
Kokanee	22%
Trout	8%
Whitefish	2%
General	35%

Table 43. Survey-wide catch rate by species estimated from angler interview data collected between April 1, 2018 and March 31, 2019 on Spirit Lake, Idaho.

Species	Total Catch Reported	Catch Rate
Black Crappie	66	0.11
Bluegill	60	0.10
Kokanee	277	0.46
Largemouth Bass	287	0.47
Pumpkinseed	10	0.02
Pygmy Whitefish	9	0.01
Rainbow Trout	39	0.06
Smallmouth Bass	52	0.09
Westslope Cutthroat Trout	25	0.04
Yellow Perch	3	<0.01

Table 44. Catch rate by species and month for fish reported by anglers from April 1, 2018 through March 31, 2019 on Spirit Lake, Idaho.

Species	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar
Black Crappie	--	0.58	--	0.01	0.00	0.02	--	--	--	--	--	--
Bluegill	--	--	0.48	--	0.12	0.08	--	--	--	--	--	--
Kokanee	--	0.59	0.47	0.25	0.40	0.19	0.02	--	--	--	2.28	--
Largemouth Bass	0.08	0.28	0.20	0.40	0.89	0.21	0.49	--	--	--	--	--
Pumpkinseed	--	--	0.11	0.04	--	--	--	--	--	--	--	--
Pygmy Whitefish	--	--	--	0.03	--	--	0.08	--	--	--	0.09	--
Rainbow Trout	0.04	0.04	0.03	0.26	0.02	0.02	0.06	--	--	--	0.16	--
Smallmouth Bass	--	0.09	0.11	0.10	0.09	0.12	0.08	--	--	--	--	--
Westslope Cutthroat Trout	0.24	0.02	--	0.04	0.01	0.02	0.04	--	--	--	0.28	--
Yellow Perch	--	--	--	0.03	--	--	--	--	--	--	0.03	--

Table 45. Catch and harvested proportion of catch by species from Spirit Lake, Idaho from April 1, 2018 through March 31, 2019.

Species	Total Catch	% Harvest
Black Crappie	3,872	51%
Bluegill	7,419	0%
Kokanee	22,899	98%
Largemouth Bass	12,071	4%
Pumpkinseed	713	0%
Pygmy Whitefish	2,299	5%
Rainbow Trout	4,241	79%
Smallmouth Bass	4,054	0%
Westslope Cutthroat Trout	1,619	58%
Yellow Perch	206	42%

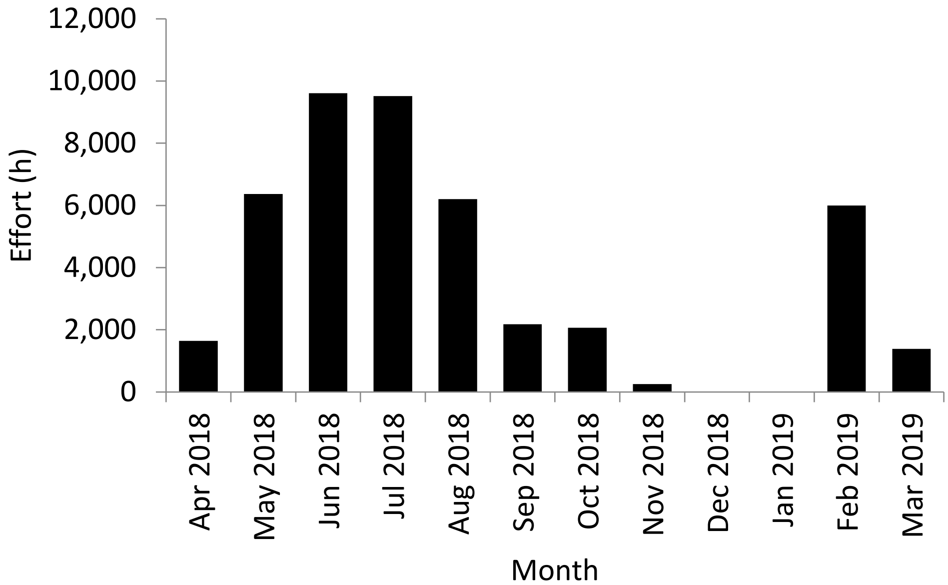

Figure 61. Angler effort by month estimated from Spirit Lake, Idaho from April 2018 through March 2019.

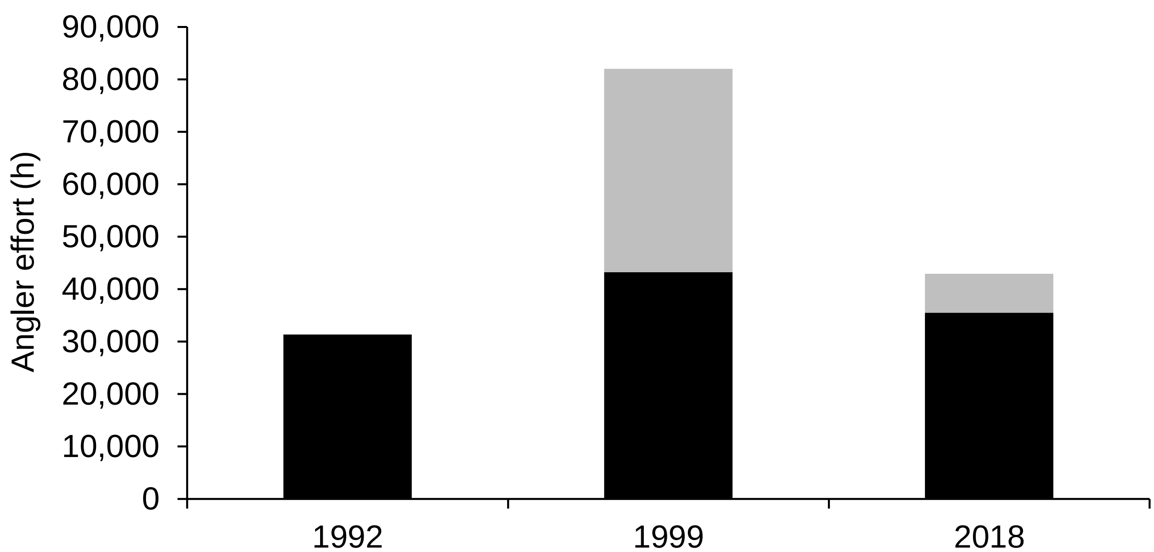

Figure 62. Comparison of angler effort estimated during January 1 - March 31 (gray) and April 1–September 30 (black) in Spirit Lake during creel surveys in 1992, 1999, and 2018. Creel surveys were not conducted during January 1 - March 31 in 1992.

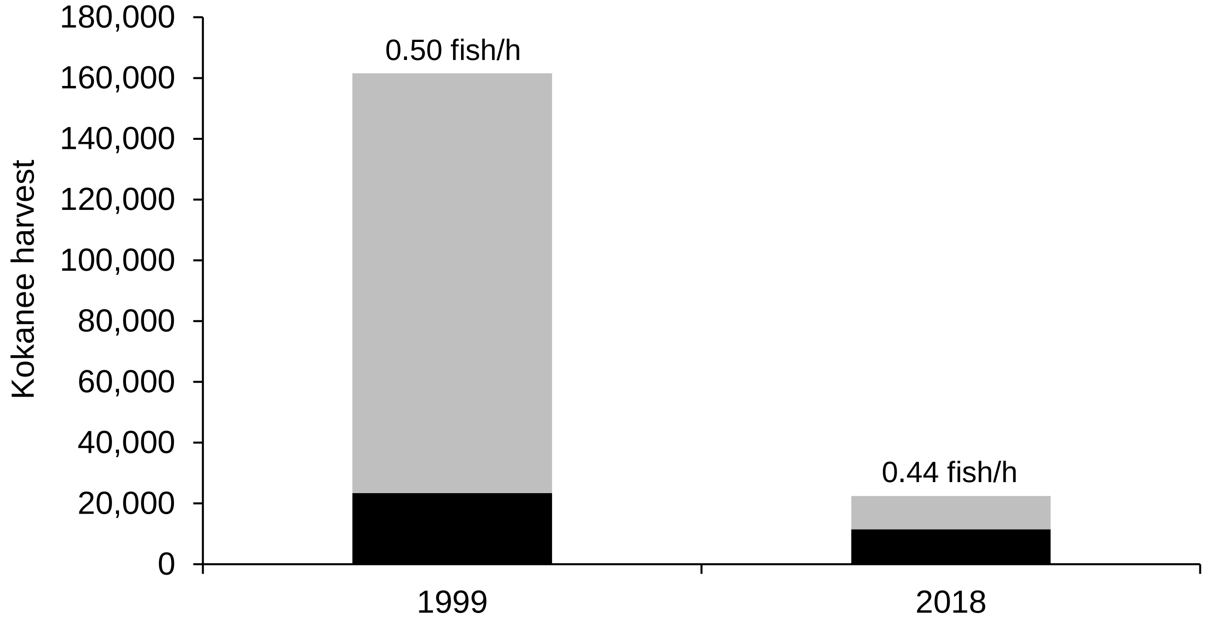

Figure 63. Estimate of kokanee harvest in Spirit Lake during January – March (gray) and April - September (black) in the 1999 and 2018 creel surveys. Data labels indicate kokanee catch rates during each of those years to provide comparison context.

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